# UrbanHabitats

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## Birds in the Urban Environment

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# UrbanHabitats

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Page 1 **Nesting Success and Life-History Attributes of Bird Communities Along an Urbanization Gradient**
Joseph A. Reale[1] and Robert B. Blair[2]
[1] City of Boulder Open Space and Mountain Parks, P.O. Box 791, Boulder, CO 80306
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Page 25 **Urban Bird Diversity as an Indicator of Social Diversity and Economic Inequality in Vancouver, British Columbia**
Stephanie J. Melles
Department of Zoology, University of Toronto, Toronto, Ontario, Canada, M5S 3G5

Page 49 **Microhabitat Selection and Singing Behavior Patterns of Male House Finches (*Carpodacus mexicanus*) in Urban Parks in a Heavily Urbanized Landscape in the Western U.S.**
Esteban Fernández-Juricic, Rachael Poston, Karin De Collibus, Timothy Morgan, Bret Bastain, Cyndi Martin, Kacy Jones, and Ronald Treminio
Department of Biological Sciences, California State University, Long Beach, 1250 Bellflower Blvd., Long Beach, CA 90840-3702.

Page 70 **Using Citizen Science in Urban Bird Studies**
Rachel E. McCaffrey
School of Natural Resources, University of Arizona, Tucson, AZ 85721

Page 87 **Avian Response to Restoration of Urban Tidal Marshes in the Hackensack Meadowlands, New Jersey**
Alison Seigel,[1] Colleen Hatfield,[2] and Jean Marie Hartman[3]
[1]Rutgers University, Graduate Program in Ecology and Evolution, 1 College Farm Road, New Brunswick, NJ 08901
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New Brunswick, NJ 08901)
[3] Rutgers University, Department of Landscape Architecture, 93 Lipman Drive, New Brunswick, NJ 08901

Page 117 **History of the Eastern Screech-Owl (*Megascops asio*) in New York City, 1867–2005**
Robert DeCandido
Hawk Mountain Sanctuary, Acopian Center for Conservation Learning, 410 Summer Valley Road, Orwigsburg, PA 17961

## ALSO IN THIS ISSUE...

Page 134 **Bringing the Urban Environment Into the Classroom: Learning From an Estuarine Mesocosm**
James C. Sullivan,[1] Theresa O'Neill,[2] and John R. Finnerty[1]
[1] Biology Department, Boston University, 5 Cummington Street, Boston, MA 02135
[2] Odyssey High School, South Boston, MA

Page 158 **Benthic Communities in *Spartina alterniflora*– and *Phragmites australis*–Dominated Salt Marshes in the Hackensack Meadowlands, New Jersey**
Catherine E. Yuhas,[1] Jean Marie Hartman,[2] and Judith S. Weis[3]
[1] New Jersey Sea Grant College Extension Program, New York–New Jersey Harbor Estuary Program Office, 290 Broadway, 24th Floor, New York, NY 10007 (address at time of research: New Jersey Institute of Technology/Rutgers University, Federeated Graduate Program in Environmental Science, Newark, NJ 07102)
[2] Rutgers University, 93 Lipman Drive, New Brunswick, NJ 08901
[3] Department of Biological Sciences, Rutgers University, 411 Boyden Hall, Newark, NJ 07102

Page 192 **The Changing Flora of the New York Metropolitan Region**
Steven E. Clemants and Gerry Moore
Brooklyn Botanic Garden, 1000 Washington Avenue, Brooklyn, NY 11225

# Nesting Success and Life-History Attributes of Bird Communities Along an Urbanization Gradient*

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## Abstract

The increase in urbanization in North America has raised concerns regarding impacts on avian populations. In this study, we measured the nesting success of American robins and northern cardinals and analyzed the changes in bird community along an urbanization gradient in southwestern Ohio. We found that nesting failure was not significantly correlated with the gradient, but that it was correlated to nest height, which decreased significantly from the most natural to the most urban sites. We also found that nesting failure was not predicted by the density of adult birds. At the community level, the number of species that use a multiple-brood breeding strategy increased with urbanization. Furthermore, birds identified as high-nesting species reached their highest proportion at the most natural sites and decreased in number with urbanization. In contrast, low-nesting species exhibited the reverse trend. These findings suggest that nesting success—determined by nest site availability and the ability to produce multiple broods—may drive the distribution of avian species along an urbanization gradient, and that nesting site is a critical resource that regulates the distribution of birds in urban environments.

**Keywords:** American robin; community; life histories; nesting success; northern cardinal; southwestern Ohio; urbanization

## Introduction

The growth of urban centers in the United States has profound effects on natural ecosystems. Increases in urban populations result in the wholesale conversion of agricultural and forest tracts into urban and suburban environments. The result of this change in land use is a mosaic of land types ranging from entirely built-up urban centers to natural or seminatural areas (McDonnell, Pickett & Pouyat, 1993). Land use for human purposes alters both the structure and function of ecosystems and is the leading cause

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of biological diversity loss worldwide (Vitousek, Mooney, Lubechenco & Melillo, 1997).
As development reaches into rural areas, many forests, if not fragmented or obliterated outright, are enveloped by human settlement (Friesen, Eagles & Mackay, 1995). This has imposed great stress on avian populations, with many songbird species experiencing declines in some portion of their range (Wilcove & Terborgh, 1984; Askins, Philbrick & Sugeno, 1990; Sauer, Hines & Fallon, 2005). As breeding habitat becomes more fragmented, nest predation increases (Gates & Gysel, 1978; King, Griffin & DeGraaf, 1996; Bayne & Hobson, 1997), brood parasitism increases (Brittingham & Temple, 1983), interspecific competition for resources is more pronounced (Cawthorne & Merchant, 1980; Ambuel & Temple, 1983), and pairing success decreases (Gibbs & Faaborg, 1990; Villard, Martin & Drummond, 1993).

The effects of urbanization on bird communities are well documented (Hoover, Brittingham & Goodrich, 1995; Friesen et al., 1995; Blair, 1996; Morse & Robinson, 1998; Porneluzi & Faaborg, 1999). These studies show that total and native species richness decline at high levels of development. Individual species, however, display differing responses to urbanization. Some birds reach peak densities in urban or suburban settings, while others reach peak densities at natural sites (Mills, Dunning & Bates, 1989; Blair, 1996; Clergeau, Savard, Mennechez & Falardeau, 1998; Gering & Blair 1999).

The cumulative response of individual species to urbanization also results in changes at the level of the bird assemblage. Blair (2001) examined the distribution and abundance of birds along an urban gradient in southwestern Ohio. This study included a spectrum of habitat types created by urbanization, ranging from a pristine nature reserve to a highly developed urban center. Individual species displayed patterns of abundance along the gradient that reflect their level of tolerance for human impact. For example, European starlings (*Sturnus vulgaris*) were labeled "urban exploiters" based on their higher abundance at the urban end of the gradient. On the opposite end, ovenbirds (*Seiurus aurocapilla*) were labeled "urban avoiders" based on their high abundance at the natural end of the gradient and their complete absence from the urban end.

The urban bird community is most strongly influenced by vegetation, with the volume of native vegetation being most closely correlated with native bird density and species richness (Mills, Dunning & Bates, 1991). The urban environment favors species that can utilize small, discontinuous patches of vegetation (Beissinger & Osborne, 1982), and densities of urban exploiters are strongly correlated with lawn area and the volume of exotic vegetation (Mills, Dunning & Bates, 1989). The relationship between habitat variables such as vegetation density and species diversity has traditionally been explained in terms of food abundance and foraging niche space (MacArthur, 1961; MacArthur, MacArthur & Preer, 1962;

MacArthur, Recher & Cody, 1966; Martin & Karr, 1986). However, Martin (1988b) hypothesized that species distribution may also be influenced by the availability of suitable nesting sites.

Nest predation is the most common cause of nesting failure among open-cup nesting passerines (Ricklefs, 1969; Martin, 1988a). As a result, predation pressure may be an important factor in regulating densities and distributions of birds (Emlen, 1974). The influence of nest predation at the level of assemblage has been largely unstudied (Martin, 1988c). Because the intensity of nest predation varies with attributes of the nesting substrate and nest height (Ricklefs, 1969; Martin & Roper, 1988; Martin & Li, 1992; Martin, 1993a), the effects of vegetation on species distributions may be due in part to the availability of suitable nesting sites (Martin 1988c).

Many researchers have studied the effects of grazing, clear cutting, and other types of habitat alteration on nest predation (for example, see Wilcove, 1985; Hoover et al., 1995; Bayne & Hobson, 1997; Ammon & Stacey, 1997; Zanette & Jenkins, 2000). Recently, ecologists have started investigating these effects in urban settings. Any changes in the assemblage of predators coinciding with increased urbanization (Tomialojc, 1970; Churcher & Lawton, 1987) would be expected to change predation pressure, which in turn may affect overall community structure.

The most common way of assessing community-level predation pressure in birds is by using artificial nests. To measure changes in predation pressure associated with urbanization, Gering and Blair (1999) placed handmade nests containing Japanese quail eggs (*Coturnix coturnix*) in six sites representing a gradient of urbanization. They found that the overall frequency of nest predation decreased with increasing levels of urbanization. This trend was also supported by a study on predation of actual robin nests (Morneau, Lepine, Decarie & Desranges, 1995). From these studies, they proposed the "predatory relaxation" hypothesis, in which urban environments serve as safe zones with reduced predation pressure because of fewer predators.

Alternatively, Sasvari, Csorgo, and Hahn (1995) measured the predation rate on artificial nests in an urban park and contrasted it with the predation rate in a mixed oak-beech forest. They concluded that reduced species richness of birds in urban environments was a result of increased predation pressure. Their results were supported by Jokimaki and Huhta (2000), who also measured predation rates in a series of parks with different levels of surrounding urbanization. However, based on beak impressions left on plasticine eggs, most of the predation in the town center was attributed to avian nest predators. Willebrand and Marcstrom (1988) demonstrated that birds are the primary predators on artificial nests and that mammals are more often found to prey on real nests. As a result, artificial nests do not always reflect true differences in nest predation among different nest sites and heights (Martin, 1987).

Clear differences in predator response to artificial nests and a lack of correlation between artificial-nest and real-nest predation probabilities are shown in numerous studies (see Major & Kendal, 1996, for a review, and King, DeGraaf, Griffin & Maier, 1999). Furthermore, artificial nests only model predation on the incubation stage of nesting. But nesting failure can be attributed to multiple causes other than predation in the incubation stage, including predation of hatchlings, abandonment due to adult mortality or lack of resources, and destruction of nests through natural catastrophes such as wind or rain. Brood parasites, such as the brown-headed cowbird (*Molothrus ater*), can also negatively affect the outcome of nesting attempts, and their impact may vary greatly with habitat attributes (Hoover, Brittingham & Goodrich, 1995).

As a result, conclusions about community-level effects of nest failure based on artificial nests may be spurious. To assess the effects of urbanization on nesting failure and related impacts on community organization, data on real nests are needed. The goal of this study was to measure nesting success of real nests and to assess the relationship between nesting outcome and the avian assemblage organization along an urban gradient in southwestern Ohio. To do this, we measured nesting success of real nests and analyzed the composition of bird community along the urban gradient.

Specifically, we addressed the following four questions:

1. Does increased urbanization affect the nesting success of urban birds?
2. Do nest-site characteristics influence nesting success, and if so, do these characteristics vary across a gradient in urbanization?
3. Does the reproductive strategy of birds change with increasing levels of urbanization?
4. Is the change in the bird community associated with urbanization correlated with a change in nesting guilds?

## Methods

### Study Sites

We used an urban gradient previously established by Gering and Blair (1999). It was composed of six 16-hectare sites representing different levels of urbanization. Four of the sites were located within Oxford, Ohio, a small city (14.7 square kilometers) with a population of 21,943 (U.S. Census Bureau, 2000). The other sites—a golf course and a nature preserve—were located seven kilometers north of Oxford. The sites of the urban gradient, listed in order of increasing urbanization, were as follows:

1. Nature preserve site—Hueston Woods Nature Preserve

   This site was located in a 67-hectare woodlot (Hueston Woods) composed of mature beech-maple forest, with the exceptions of a small parking area, narrow service road, and moderate-use foot trails. The area is encompassed by a state park (975 hectares), which consists of young and mature deciduous forest, old fields, infrequent stands of conifers, and a 250-hectare lake.

2. Open-space site—Peffer Memorial Park

   This site was located in an 80-hectare park that is managed by Miami University, Oxford, as a multiple-use trail system for local residents. The university purchased the park as two separate parcels, a pastureland and an agricultural field, in 1955 and 1966, respectively. The current vegetation is predominantly secondary growth of low stature. The park can be considered a "wildland" because buildings and pavement are absent; however, it has a history of substantial human manipulation.
3. Golf course site—Hueston Woods Golf Course

   This site was located in a 102-hectare golf course seven kilometers from Oxford. The course was established in 1980. Its rough consists primarily of native trees (e.g., maple, beech) and grasses. Less than 2% of its area is covered with buildings and pavement.
4. Residential district site—Oxford

   This 16-hectare site covered approximately ten city blocks within a much larger residential area and mainly consisted of single-family houses built between 1960 and 1974. The vegetation is largely dense lawns, gardens, shade trees, and both native and ornamental flora (Beissinger & Osborne, 1982). The site is suburban, with approximately 32% of its area covered with buildings and pavement.
5. Apartment complex site—Oxford

   This 16-hectare site consisted of a series of multilevel condominium-style apartment complexes with interspersed parking lots, isolated trees, and small landscaped areas. These complexes were developed between 1960 and 1980 and are surrounded by a residential area on three sides and a commercial area on the fourth side. The site is marginally urban, with approximately 61% of its area covered with buildings and pavement.
6. Business district site—Oxford

   This 16-hectare site covered an expanse of four city blocks by two city blocks originally constructed in the mid-1800s, and it is surrounded by an older residential area. It is a distinctly urban site, with approximately 81 percent of the area covered in buildings and pavement. The site is composed primarily of low-rise office buildings, parking lots, ornamental shrubs, and landscape trees.

We applied principal components analysis (PCA) using PC-ORD software (Version 4, 1999) to assign each site a value denoting its level of urbanization. Landscape-level estimates of the percent of cover of trees and shrubs, lawn, buildings, and pavement for each of the sites had been previously obtained (Gering & Blair, 1999).

A combination of ortho-digital maps and aerial photographs were used to quantify surface cover. The land cover for each of the sites was broken down into the percent of surface area covered by buildings, pavement, lawn or grassland, and trees or shrubs. The estimates of percent cover were analyzed with PCA, and the scores for each site from the first axis of the analysis were used to assign site values as follows: business district (2.48); apartment complex (1.75); residential district (0.30); golf course (–0.84); and open space (–1.70). We used this approach to create a continuous measure of urbanization of the sites so that we would not be limited to statistical analyses based on rank order alone.

### Study Species

Using abundance data based on Gering and Blair (1999) and Blair (2001), we identified the American robin (*Turdus migratorius*) and northern cardinal (*Cardinalis cardinalis*) as two species that were present at most sites along the gradient. These species displayed differing patterns of abundance, with robins being more urban tolerant (appearing at higher densities at the urban end of the gradient) and cardinals showing a preference for the natural end of the gradient (Figure 1).

### Nest Location and Fate

We located nests during May and June 2000 by thoroughly searching each 16-hectare site and observing behavioral cues of parental birds, including nest building and vocalization. We marked nest sites by placing flagging within 15 meters of the nests and recording their position with GPS. After locating these nests, we used a convex mirror mounted on a ten-meter expandable pole (Parker, 1972) to determine the contents and to minimize contact with nests.

We discovered all nests during the building, egg laying, or early incubation stages. We monitored nests every three to four days during incubation to determine hatch dates and, after hatching, to determine fledging success. For each nest, we recorded the number of eggs laid, hatched, and fledged as well as any predation events or other instances of egg loss.

### Nest Characteristics

After each nesting attempt was completed, we measured habitat characteristics at and around the nest. We recorded the height of the nest and its depth into the vegetation. We used the diameter in centimeters at breast height (DBH) of the nesting vegetation to classify the nest location as a tree (DBH > 10 cm), sapling (10 cm > DBH > 5 cm), or shrub (DBH < 5 cm). We measured the distance from the nest to the nearest tree (DBH > 10 cm) and the nearest edge (defined as an area of open canopy).

### Community Classification

Gering and Blair (1999) had previously recorded the species present and relative abundance of birds at each site. Using these data, we classified each species according to:

a) breeding strategy (single, double, or multiple broods per season)

b) nesting height in meters (low: 0 to 3; mid-height: 3 to 6; and high: > 6)

c) nest location (tree, shrub/ground, or other—cliffs, buildings, etc.)

d) nest type (open or cavity)

We based these classifications on the information in Ehrlich, Dobkin, and Wheye (1988).

### Statistical Analyses

*Nesting success of robins and cardinals*. We analyzed each species separately for the dependency of nest fate (unsuccessful = 0, successful = 1) on site using a logistic regression model (Collett, 1991). The model was specified as follows: $\text{Logit}(p) = ß_0 + ß_1(\text{site}) + ß_2(\text{species}) + ß_3(\text{site}) \times (\text{species})$, where p is the proportion of nests that failed ($0 < p < 1$), with $\text{Logit}(p) = \ln(p/p + 1)$. "Site" is a continuous variable for site based on the PCA value. "Species" is an indicator variable that discriminates between species (i.e., species = 0, for cardinals, and species = 1, for robins).

We used a separate logistic regression model to determine if species density predicted nest fate. The model was specified as follows: $\text{Logit}(p) = ß_0 + ß_1(\text{density})$, where p is the proportion of nests that failed ($0 < p < 1$), with $\text{Logit}(p) = \ln(p/p + 1)$. Density is the density of the species being tested.

We further analyzed each species for dependency of nest fate on nest characteristics. The full model was specified as $\text{Logit}(p) = ß_0 + ß_1(H) + ß_2(D) + ß_3(E) + ß_4(T) + ß_5(V)$, where p is the proportion of nests in the stage being tested that failed ($0 < p < 1$), with $\text{Logit}(p) = \ln(p/p + 1)$. The letters H, D, E, and T are continuous variables for nest height, depth of nest in vegetation, edge distance, and distance to the nearest tree, respectively. V is a categorical variable representing vegetation type, ($V = 1$ for shrubs, $V = 2$ for saplings, and $V = 3$ for trees).

Starting with the full model, we used Wald's statistic (W) to identify significant variables ($p < 0.05$). We eliminated variables stepwise from the model until only significant variables remained. Using a one-way ANOVA, we tested variables that were significant in predicting nesting success for differences in their mean values across sites. We also used linear regression models to determine whether nest characteristics that were significant predictors of nesting success were related to the degree of urbanization.

*Nesting guilds.* We analyzed the community classifications using a chi-square ($X^2$) test for independence (Samuels & Whitmer, 1999). The null hypothesis for all comparisons ($H_0$) was that the proportion of individuals with a given nesting characteristic was equal to the proportion of individuals in the entire population with that trait (i.e., assemblages were random). We calculated expected values by multiplying the number of species present at a given site by the relative proportion of all species that possessed the trait being tested. For example, 30 out of 42 total species (71.5%) were open-nesting species. The nature preserve had 17 species present, so we expected that 71.5%, or 12.14 species, would be open nesting. We used community characteristics that varied significantly across sites in a linear regression model to determine if they were significantly correlated to the degree of urbanization.

## Results

### Nesting Success

We discovered nests in five of the six sites. At the nature preserve, robins were absent and cardinals were present in very low numbers and restricted to a small portion of the preserve. Therefore, we did not include data on nesting success from the nature preserve.

We discovered 85 nests in the five remaining sites: 51 robin nests and 34 cardinal nests. Across all sites, 65% of robin nests and 71% of cardinal nests were successful (Table 1). Density was not a significant predictor of nest fate for robins (DF = 1, W = 0.08, $p = 0.67$) or cardinals (DF= 1, W = 0.01, $p = 0.84$). The site was also not a significant predictor of nest fate for robins (DF = 1, W = 0.66, $p = 0.42$) or cardinals (DF= 1, W = 0.002, $p = 0.96$).

Nest height was the only significant predictor of nest success for both species (robins: DF = 1, W = 14.157, $p < 0.01$; and cardinals: DF = 1, W = 9.410, $p < 0.01$). In both species, higher nests were more likely to be successful.

Mean nest height for robins and cardinals differed significantly across sites (robins: DF = 4, F = 14.62, $p < 0.001$; cardinals: DF = 4, F = 4.94, $p = 0.03$). The mean nest height between the two species was significantly different (DF = 1, F = 10.06, $p = 0.002$). For both species, the mean nest height decreased with increasing levels of urbanization (robins: $R^2 = 0.23$, $p < 0.001$; cardinals: $R^2 = 0.13$, $p = 0.03$) (Figure 2).

### Nesting Guilds

Blair (2001) detected 43 species across all six sites of the urban gradient. We eliminated brown-headed cowbird from the guild analysis because of its unique status as a brood parasite (see Table 2 for species classifications). We classified 29% of the species as high nesting, 33% as mid-height nesting, and 37% as low nesting. Tree-nesting species accounted for 55% (n = 23), and shrub/ground-nesting species made up 31% (n = 13). We classified an additional 14% (n = 6) as having some other nesting location. The population was 55% (n = 23) single-brooding, 24% (n = 10) double-brooding, and 21% (n = 9) multiple-brooding species.

The brooding strategy of species differed significantly across sites (DF = 10, $X^2 = 61.12$, $p < 0.001$). The proportion of single-brooding species declined dramatically with increased urbanization. Double-brooding species peaked in the open-space area and were entirely absent from the most urban sites. There was a strong trend toward an increasing proportion of species using a multiple-brood strategy with increased urbanization. The number of broods a species attempts in a year was related to the degree of urbanization ($R^2 = 0.93$, $p < 0.01$) (Figure 3).

The nesting-height guild differed significantly across sites (DF = 10, $X^2 = 27.73$, $p < 0.01$). The proportion of species using high nest sites peaked at the most natural sites and declined with increasing urbanization, with no high-nesting species present in the two most urban sites. The proportion of mid-height-nesting species reached peak values at the moderately urbanized sites (residential district and golf course) while remaining relatively constant at the extreme ends of the gradient. Low-nesting species exhibited peak proportions at the open-space site and showed a general increase from the golf course to the business district. There was a marginally significant inverse correlation between nest height and the degree of urbanization ($R^2 = 0.96$, $p = 0.06$) Figure 4).

Nesting location did not differ significantly across the sites (DF = 10, $X^2 = 15.64$, $p = 0.09$). Tree-nesting species were evenly distributed across sites. The proportion of shrub- and ground-nesting species was greatest at the open-space site but was otherwise evenly distributed. However, none of the species with unique nesting locations was present at the least urban end of the gradient, and their relative numbers increased with increasing urbanization (Figure 5). The distribution of cavity-nesting and open-nesting species did not differ significantly across sites (DF = 5, $X^2 = 5.91$, $p = 0.11$).

## Discussion

Many factors may act as selective forces in determining the nesting outcome, habitat selection, and community structure of birds. These include the availability of adequate forage (Lack, 1954; Martin, 1987), interspecific competition for resources (Holmes, Sherry & Sturges, 1986; Moulton & Pimm, 1986), and the availability of vegetation and habitat, which provides cover and nesting or feeding substrate (Karr & Roth, 1971; Mills, Dunning & Bates, 1991). In urban ecosystems, the abundance and diversity of vegetation and habitat heterogeneity affect the species richness and diversity of birds (Lancaster & Rees, 1979; Dowd, 1992). Local features are more important than landscape-level features in determining the composition of urban-bird communities (Clergeau et al., 1998).

In our study, the first question we asked was whether increasing levels of urbanization influenced the nesting success of birds. Although nesting success varied widely for robins (43% to 73%) and cardinals (56% to 78%), the variation was not attributable to location along the urban gradient. Furthermore, the relative abundance of each species was not predicted by nesting success. Our findings fail to support the conclusion that nesting success influences the relative density of individual species. This means that individuals within each species are probably unable to select nesting sites based on the suitability of the landscape in terms of nesting success. Further, this result has implications concerning the use of density measurements to indicate habitat quality. These findings support the review by Bock and Jones (2004) that bird densities generally are good predictors of nesting success except in human-dominated environments.

However, in answer to the second question of our investigation, nest height significantly influenced nesting outcome for both species, with higher nests being more likely to succeed. At the same time, mean nest height decreases with urbanization. Consequently, as sites become urbanized, individual birds are forced to choose nesting locations that decrease their chances of success. This may be due to a lack of available nesting sites.

Third, we asked if reproductive strategy changed with urbanization. While neither of our study species—robins and cardinals—showed any difference in nesting success across the gradient, the proportion of species in the entire bird community that rely on a strategy of multiple brooding dramatically increased with urbanization. Both robins and cardinals are suburban-adaptable species that actively defend their nests, so they may not adequately represent the entire avian assemblage. Single-brooding species show a strong preference for the most undisturbed sites, and their numbers decrease with urbanization. This shift in the overall breeding strategy of the community is due, in large part, to a loss of migratory species in more urban sites. Multiple-brooding species may be able to compensate for nesting losses in habitats with low nesting success, while most neotropical migrants are restricted to one or two broods per season because of the energy costs associated with long-distance migration (Whitcomb, Robbins, Lynch, Klimkiewicz & Bystrak, 1981; Terborgh, 1989).

The change in breeding strategy from low productivity (single brood) in undisturbed sites to high productivity (multiple broods) in urban sites may indicate a change in overall nesting-habitat quality. The strategy change may imply that increased urbanization results in a decrease in nesting success. Species that are able to afford the loss of one or more broods per season may be better adapted for urban sites, while those that rely on a single brood each year may be unable to maintain populations in urban areas.

Consistent with the findings of other studies (e.g., Ricklefs, 1969; Skutch, 1985; Martin, 1988b), nest predation was the primary cause of nesting failure for all sites. Because of the potentially enormous influence of predation on avian assemblages, the community-level influences of nest predation have become a primary focus in avian studies. It has been hypothesized that avian community organization may be regulated by the availability of nest sites that minimize the risk of predation (Martin, 1992). The results of this study suggest that nesting success over the entire breeding season rather than of individual nests may drive the distribution of avian species in urbanizing environments.

The variation in the avian assemblage that occurs in conjunction with variations in vegetation has been attributed to differences in foraging strategy (MacArthur, 1961; MacArthur, MacArthur & Preer, 1962; MacArthur, Recher & Cody, 1966; Sabo & Holmes, 1983). However, the availability of suitable nesting sites may be more limiting than food (Rosenberg, Terrill & Rosenberg, 1987). Most birds are highly specialized in their nesting-site location, while foraging preferences are more generalized and exhibit greater interspecific overlap (Martin, 1988a; Martin, 1993b). After comparing residential neighborhoods of Oxford and the nature preserve, Beissinger and Osborne (1982) concluded that the vertical distribution of vegetation could not explain the decrease in avian species diversity associated with urbanization. However, the total volume of vegetation was significantly lower in the residential area, and this may have been a factor limiting food availability, cover, and nest placement. The changes in nesting-height guild across the urban gradient in our study are consistent with the hypothesis that the availability of suitable nesting sites is a selective factor in determining the composition of the avian bird assemblage in urbanizing environments.

At the most urban end of the gradient, low-nesting species completely replace high-nesting species. The transition from undisturbed woodland to a highly modified business district results in a shift in vegetation from tall, native trees and sparse understory vegetation to short, isolated ornamental trees and shrubs (Beissinger & Osborne, 1982; Gering & Blair, 1999). This should favor species that can make use of lower nesting sites. In addition, the introduction of novel nesting locations (i.e., chimneys, dryer vents, rain gutters, and other man-made structures) allows species with unique or flexible nesting preferences such as starlings, house sparrows (*Passer domesticus*), chimney swifts (*Chaetura pelagica*), and rock doves (*Columba livia*) to inhabit highly urban sites.

The results of this study indicate that nesting site is a critical resource that regulates the distribution of birds in an urban environment. The habitat alteration that accompanies urbanization reduces the diversity and availability of nesting sites. This may force individuals to use poor-quality nest sites, and our data suggest that this may alter the species of birds in the community. It may be possible to increase the numbers of native species in urban settings by using more native plants in the landscape and increasing the volume of vegetation that provides optimal nesting sites.

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## Glossary

**ANOVA (Analysis of variance):** Statistical method that yields values that can be tested to determine whether a significant relation exists between variables.

**Brood parasite:** An animal that lays its eggs in the nest of a member of its own or another species.

**Chi-square ($X^2$) test:** A statistical test for assessing the significance of departures of sets of whole numbers (those observed) from those expected by the hypothesis.

**Community structure:** The physical and biological components of a community, defined as an association of interacting populations of species.

**Interspecific competition:** Competition for resources between different species.

**Linear regression model:** A statistical model used to estimate the conditional expected value of one variable y given the values of some other variable or variables x.

**Logistic regression model:** A statistical model used to predict a discrete outcome, such as group membership, from a set of variables that may be continuous, discrete, dichotomous, or a mix of any of these.

**Multiple-brood breeding strategy:** The production of more than one group of young per breeding season.

**Nesting guild:** A group of species exploiting similar nesting resources in a similar fashion.

**Niche space:** A multidimensional suite of environmental factors (or "space") that affects the welfare of a species.

**Null hypothesis (H0):** A hypothesis that states the converse of the expected results in an experiment.

**Open-cup nest :** An open cup-shaped nest made with a variety of materials such as grass, moss, lichen, or spider web.

**Ortho-digital maps:** Digital maps created using orthophotography, which employs simple or differential rectification to remove displacements from conventional-perspective photography caused by camera tilt and relief of terrain.

**Principal components analysis (PCA):** A multivariate analysis technique that orders a set of objects in any number of dimensions (fewer is better). It involves Eigen analysis of a correlation matrix.

**Wald's statistic:** The result of a statistical test to compare two correlated proportions.

**Figure 1. Mean daily density (birds/hectare ± SE) across sites presented in order of increasing urbanization for (a) American robins and (b) northern cardinals.**

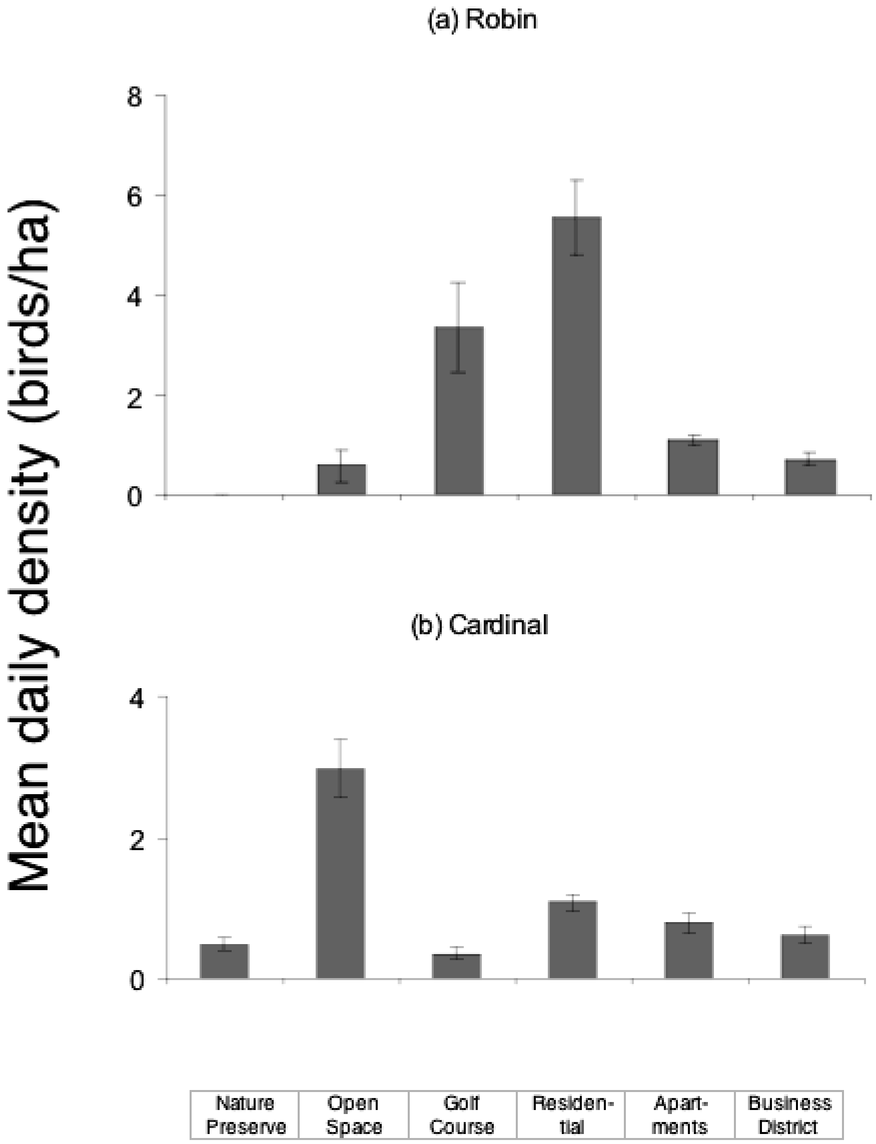

**Figure 2. Mean (± SE) for nesting height of American robins and northern cardinals across sites presented in order of increasing urbanization.**

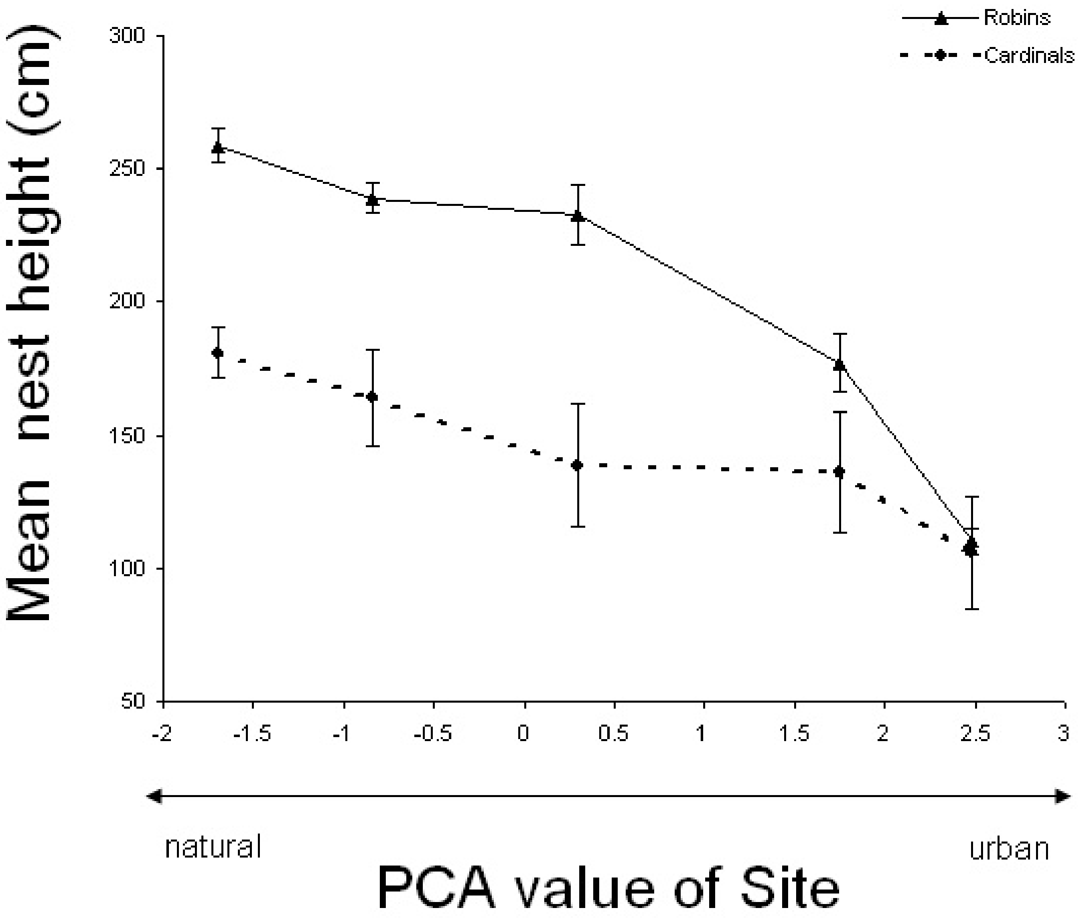

**Figure 3. The proportion of species present in each site presented in order of increasing urbanization for (a) single brooders, (b) double brooders, and (c) multiple brooders.**

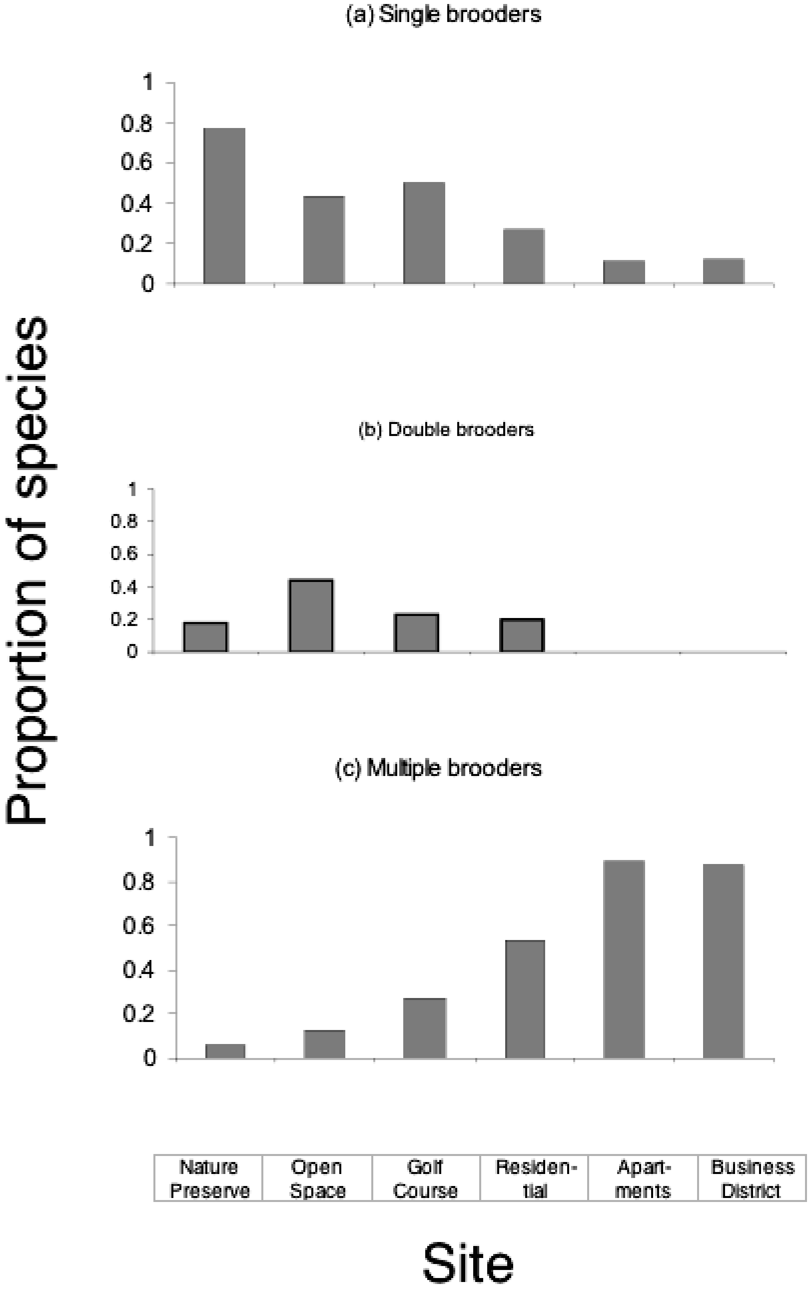

**Figure 4. The proportion of species present in each site presented in order of increasing urbanization for (a) high-nesting, (b) mid-height nesting, and (c) low-nesting species.**

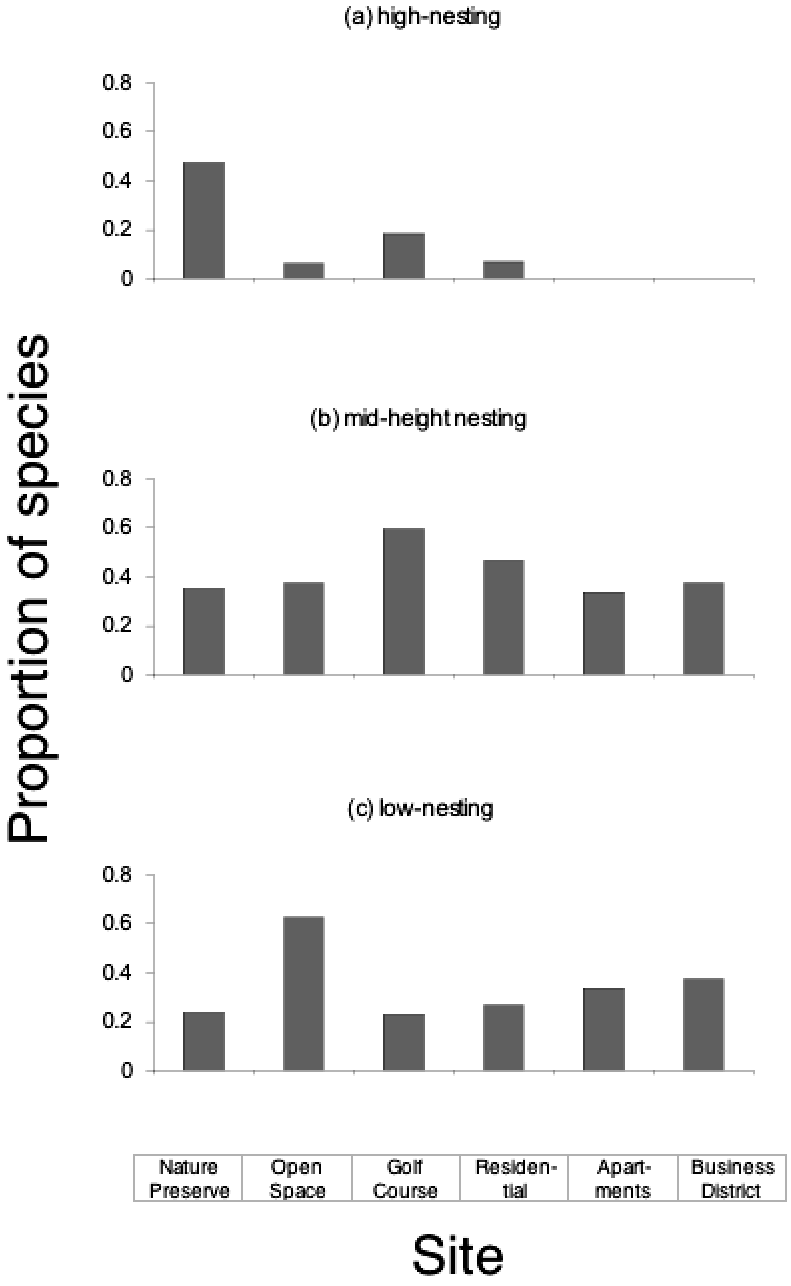

**Figure 5. The proportion of species present in each site presented in order of increasing urbanization, for (a) tree-nesting species, (b) shrub- and ground-nesting species, and (c) other nesting locations.**

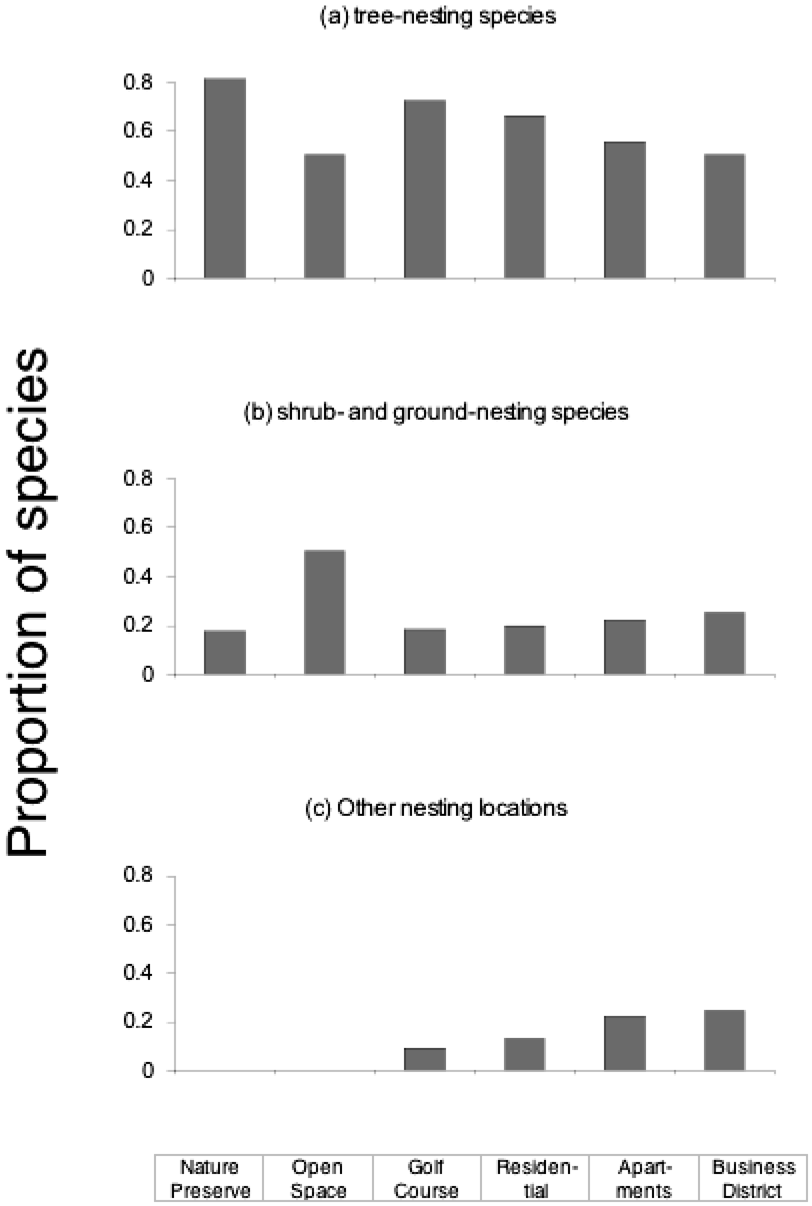

**Table 1. Total number of nests and the percentage of them that were successful within each site for American robins and northern cardinals. Sites presented in order of increasing urbanization.**

| | **Nest type** | | | |
|---|---|---|---|---|
| | **American robin** | | **Northern cardinal** | |
| Site | n | % successful | n | % successful |
| **Open Space** | 11 | 66.6 | 8 | 62.5 |
| **Golf Course** | 15 | 73.3 | 9 | 55.5 |
| **Residential District** | 10 | 60.0 | 6 | 77.7 |
| **Apartment Complex** | 8 | 62.5 | 6 | 66.7 |
| **Business District** | 7 | 42.9 | 5 | 60.0 |

**Table 2. Guild classification and sites found by species**

| Species | Broods | Nest height | Nesting substrate | Nest Type | Sites* |
|---|---|---|---|---|---|
| Kentucky Warbler<br>***Oporornis formosus*** | single | low | tree | open | np |
| Northern Parula Warbler<br>***Parula americana*** | single | high | tree | open | np |
| Great Crested Flycatcher<br>***Myiarchus crinitus*** | single | high | tree | open | np |
| American Crow<br>***Corvus brachyrhynchos*** | single | high | tree | open | np |
| White-breasted Nuthatch<br>***Sitta carolinensis*** | single | medium | tree | cavity | np |
| Ovenbird<br>***Seiurus aurocapillus*** | single | low | ground | open | np |
| Red-eyed Vireo<br>***Vireo olivaceus*** | single | medium | tree | open | np |
| Acadian Flycatcher<br>***Empidonax virescens*** | double | medium | tree | open | np |
| Red-bellied Woodpecker<br>***Melanerpes carolinus*** | single | high | tree | cavity | np,gc |
| Eastern Wood Pewee<br>***Contopus virens*** | single | high | tree | open | np |
| Downy Woodpecker<br>***Picoides pubescens*** | single | high | tree | cavity | np |
| Indigo Bunting<br>***Passerina cyanea*** | double | low | shrub | open | np,os |
| White-eyed vireo<br>***Vireo griseus*** | single | low | shrub | open | os |
| Eastern Towhee<br>***Pipilo erythrophthalmus*** | double | low | shrub | open | os |
| Prairie Warbler<br>***Dendroica discolor*** | double | low | tree | open | os |
| Yellow-breasted Chat<br>***Icteria virens*** | double | low | shrub | open | os |
| Gray Catbird<br>***Dumetella carolinensis*** | double | low | shrub | open | os |

| | | | | | |
|---|---|---|---|---|---|
| Field Sparrow<br>***Spizella pusilla*** | double | low | shrub | open | os |
| Blue-gray Gnatcatcher<br>***Polioptila caerulea*** | single | medium | tree | open | np,os,gc |
| American Goldfinch<br>***Carduelis tristis*** | single | medium | tree | open | os,gc |
| Carolina Chickadee<br>***Parus carolinensis*** | multiple | medium | tree | cavity | np,os,gc,rd |
| Northern Cardinal<br>***Cardinalis cardinalis*** | multiple | medium | shrub | open | np,os,gc,rd, ac,bd |
| Tufted Titmouse<br>***Parus bicolor*** | single | high | tree | cavity | np,os,gc |
| Red-winged Blackbird<br>**Agelaius phoeniceus** | multiple | low | shrub | open | gc |
| Baltimore Oriole<br>***Icterus galbula*** | single | medium | tree | open | gc |
| Eastern Kingbird<br>***Tyrannus tyrannus*** | single | medium | shrub | open | gc |
| House Wren<br>***Troglodytes aedon*** | double | medium | tree | cavity | gc |
| Barn Swallow<br>***Hirundo rustica*** | double | high | other | open | gc |
| Warbling Vireo<br>***Vireo gilvus*** | single | high | tree | open | gc |
| Chipping Sparrow<br>***Spizella passerina*** | double | low | tree | open | gc,rd |
| Blue Jay<br>***Cyanocitta cristata*** | double | medium | tree | open | gc,rd |
| Carolina Wren<br>***Thryothorus ludovicianus*** | multiple | low | tree | cavity | os,rd |
| Cedar Waxwing<br>***Bombycilla cedrorum*** | single | medium | tree | open | gc,rd |
| American Robin<br>***Turdus migratorius*** | multiple | medium | tree | open | os,gc,rd,ac, bd |
| Mourning Dove<br>***Zenaida macroura*** | multiple | medium | tree | open | gc,rd,ac,bd |

| | | | | | |
|---|---|---|---|---|---|
| Common Grackle<br>***Quiscalus quiscula*** | single | low | tree | cavity | os,rd,ac,bd |
| House Finch<br>***Carpodacus mexicanus*** | multiple | medium | other | cavity | gc,rd,ac,bd |
| Song Sparrow<br>***Melospiza melodia*** | multiple | low | shrub | open | rd,ac,bd |
| House Sparrow<br>***Passer domesticus*** | multiple | medium | other | cavity | rd,ac.bd |
| Rock Dove<br>***Columba livia*** | multiple | high | other | open | ac |
| Chimney Swift<br>***Chaetura pelagica*** | single | high | other | open | rd,bd |
| European Starling<br>***Sturnus vulgaris*** | multiple | high | other | cavity | rd,ac,bd |

**np = nature preserve, os = open space, gc = golf course, rd = residential district, ac = apartment complex, bd = business district**

# Urban Bird Diversity as an Indicator of Human Social Diversity and Economic Inequality in Vancouver, British Columbia[*]

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## Abstract

The unequal distribution of wealth in cities contributes to other forms of spatial, social, and biological inequities in complex, interacting, and self-reinforcing ways. Recent work on urban birds has often focused on community-level correlation studies of short duration in which many points along an urban gradient are surveyed for birds, and the data are related to various ecological variables measured at multiple scales. Spatial variation in urban bird communities may also reflect socioeconomic variables and cultural differences among the human population. The purpose of this paper was to examine whether socioeconomic factors (such as mean family income and ethnic diversity) also relate to the diversity and abundance of birds in Vancouver, British Columbia. I used redundancy analysis to characterize the socioeconomic gradient in a citywide study of the bird community in 44 census-defined neighborhoods. Mean family income, census tract area, and ethnicity were some of the dominant variables that correlated with most of the variation in the bird community. I found no direct relationship between neighborhood age and bird diversity and abundance. Results demonstrate that wealthier neighborhoods have more native species of birds and that these native species increase in abundance as the socioeconomic status of the neighborhood improves. With two-thirds of the world's population expected to live in cities by 2030, more and more people will grow up surrounded by a depauperate community of birds, and this could adversely affect the way people perceive, appreciate, and understand nature. Ultimately, as city birdlife diminishes and urban dwellers become dissociated from the natural diversity it represents, popular support for preserving and restoring such diversity may wane, allowing ecological conditions to further erode.

**Keywords** : biodiversity; gradient analysis; mean family income; socioeconomic variables; spatial segregation; urban ecology

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# Introduction

**"The fostering of wildife areas in cities is too complex an operation to be left in the hands of ecologists." — O.L. Gilbert**

Of the three leading causes of species endangerment (urbanization, agriculture, and interactions with nonnative species), urbanization ranks highest (Czech, Krausman & Devers, 2000; Czech & Krausman, 1997). The process of urbanization endangers species by directly replacing native habitats with development on the urban-rural fringe, and because resources in the surrounding areas are depleted to support urban economies (Czech et al., 2000). Moreover, urban areas are expected to grow substantially in coming years: By 2030, the percentage of the world's population living in urban areas is projected to increase from the current 49% to approximately 61% (United Nations, 2004). Already in the United States, up to 80% of the population lives in suburban and urban areas (Blair, 2004; Grimm, Grove, Pickett & Redman, 2001). Urban sprawl may occur even faster in developing nations currently rich in biodiversity due to improving socioeconomic conditions (Liu, Daily, Ehrlich & Luck, 2003). As a result of continuous urban encroachment on natural habitats, the majority of the earth's human population will likely be living in a state of "biological poverty" by the year 2030 (Turner, Nakamura & Dinetti, 2004).

Biological poverty occurs when urban citizens experience below-average levels of native species diversity on a daily basis (Clergeau, Mennechez, Sauvage & Lemoine, 2001; Turner et al., 2004). Research on birds in cities worldwide has been steadily accumulating, particularly over the last few decades, and results indicate that as development intensifies, bird communities become increasingly homogenized (McKinney & Lockwood, 1999). Species richness and evenness also declines in complex ways, while total bird densities increase (Campbell & Dagg, 1976; Donnelly & Marzluff, 2004; Edgar & Kershaw, 1994; Emlen, 1974; Lancaster & Rees, 1979; Turner et al., 2004). Birds are often used as a biological model because they are good ecological indicators and they are easily observable (Clergeau et al., 2001). Moreover, trends that hold for birds may hold for other species of wildlife as well.

In the past, ecologists paid little attention to urban ecosystems and focused mainly on pristine ones (Blair, 2004; Collins, Kinzig, Grimm & Fagan, 2000; Jules, 1997; Marzluff, Bowman & Donnelly, 2001; Vandermeer, 1997). But ecological studies in urban areas now seem to be on the rise (Grimm et al., 2001). Much recent work by avian ecologists has focused on community-level correlation studies of short duration, in which many points along an urban gradient are surveyed for birds (Table 1). These kinds of studies are informative and cost effective as a first step, but unless we are able to identify all the processes that generate bird-community patterns (Table 1), our efforts to influence policy and planning will be largely ineffectual (Hostetler, 2001). A lot of interesting and challenging work remains to be done.

Experimental studies that focus on the underlying biological processes that drive ecological patterns are costly and intensive—and thus often limited to examinations at relatively small spatial scales, with low sample sizes. Moreover, they tend to focus on population-level effects rather than community interactions. Hence, some combination of mensurative and experimental work, done at a variety of spatial scales, is necessary (Table 1). Furthermore, to have a greater influence on urban planning and policy, human socioeconomic factors must be integrated into our investigations of diversity in urban areas; indeed, several researchers have suggested ways to do this (Dow, 2000; Grimm et al., 2001; Grove & Burch, 1997; Hope et al., 2003; Luck & Wu, 2002; Martin et al., 2004; Marzluff et al., 2001; McIntyre et al., 2000; Pickett et al., 1997; Pedlowski et al., 2002; Turner et al., 2004; see also Table 1).

Following research by Melles, Glenn, and Martin (2003), I examined here whether socioeconomic factors related to the bird community in Vancouver, British Columbia. Specifically, I tested whether human socioeconomic variables, such as wealth and human density, related to a gradient of avian diversity and abundance in Vancouver. By examining relationships with the dominant period of house construction in a neighborhood, I also tested whether "time since disturbance" created a habitat gradient that was related to the community of birds. Finally, I examined whether there was a clear pattern of ethnic spatial segregation that corresponded to patterns evident in the bird community.

Melles et al. (2003) investigated how the bird community changed along a gradient of increasing urban development. Bird abundance and diversity were investigated in relation to ecological attributes (for example, tree cover and impervious surface cover, composition, and number of tree species) measured at multiple scales (from 50 meters to 1 kilometer around point-count stations). But ecological factors alone cannot fully explain the patterns we see in urban areas: Humans are profound and industrious agents of change, and their behaviors influence ecological processes in cities. Other studies have found significant relationships between economic status and the type and distribution of plant species planted (Hope et al., 2003; Martin et al., 2004), as well as the type and spatial distribution of street trees (Pedlowski et al., 2002). Given that birds respond to the spatial heterogeneity and distribution of vegetation (Donnelly & Marzluff, 2004), the economic status of an urban neighborhood should have some relation to its bird community.

## Methods

### Study Area

My research was conducted in the municipalities of Vancouver, Burnaby, and Coquitlam, hereafter referred to as Greater Vancouver, in British Columbia, Canada (49°18' N, 123°12' W; Figure 1). These municipalities are located within the Vancouver Census Metropolitan Area (VCMA), a land area 2,412 square kilometers in size, which includes all surrounding municipal areas (such as Burnaby, Coquitlam, Surrey, Richmond, and North Vancouver). According to population and dwelling counts conducted by Statistics Canada (1996), the VCMA has a combined population of over 1.83 million people, and the average density of individuals in the area is 7.6 people per hectare. The maximum population density in the 44 census neighborhoods examined was 262.8 individuals per hectare and the minimum density observed was 4.4 individuals per hectare. The VCMA was the fastest-growing municipal area in Canada between 1991 and 1996, showing an increase of 14.3% in the overall population, and this growth is expected to continue. The VCMA is a largely urban and suburban area interspersed with several large parks (Figure 1).

The term "urban," though quite common in everyday usage, can be somewhat subjective. Various organizations and individual researchers have defined it using such criteria as human population density, or the ratio of built (impervious surface cover) to unbuilt area (see, for example, Marzluff, Bowman & Donnelly, 2001). McIntyre, Knowles-Yánez, and Hope (2000) argue that there is a need for a working definition of urban that integrates ecological and social definitions, including baseline information about physical geography, demography, and socio-economic and cultural factors.

Here, I use Statistics Canada's census dictionary definition of the term "urban" (incidentally, I use "city" synonymously with "urban"). In this formulation, an urban area is defined as an area with a minimum population concentration of 1,000 individuals and a population density of at least 400 per square kilometer (Statistics Canada, 1996). "Urban cores" are areas with a population size greater than 100,000 individuals (i.e., Vancouver and the surrounding suburban municipalities of Burnaby and Coquitlam; see Figure 1), and "suburban areas" are defined as politically separate municipal areas located on the periphery of urban cores.

The physical geography of Greater Vancouver includes several large parks (> 3 km$^2$) transected by road and trail systems that are used by many urban residents. The parks' relatively small impervious surface cover (between 2% and 35%) primarily results from the road network.

Neighborhoods surrounding these parks have a range of population densities, mean family incomes (see Figure 2), and varying levels of impervious surface cover (from 12% at the park edge to 70% in surrounding residential areas). Impervious surface cover in suburban areas varies between 34% and 70%, with a mean of 53%.

Vancouver is a young city—in the 1880s, it was a small settlement of sawmills, houses, and forest clearings surrounded by continuously forested land (Oke, North & Slaymaker, 1992). The city has expanded to its current size over the past 125 years and is expected to continue to grow. The ocean and mountains have tended to constrain or direct Vancouver-Burnaby urban development. A consequence of development has been the complete removal of forest and ground cover. Compensating to a small degree, street trees have been planted along many city streets. Initially, the west end of Vancouver was planted with native tree species such as Douglas fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) (Oke et al., 1992). However, because they grew too fast and their root systems buckled the sidewalks, these species were eventually deemed unsuitable for urban settings. Thus, as the city grew eastward and southward, the trees planted were largely nonnative species, over a third of them nonnative cherries or plums (*Prunus* species). The result was an uneven distribution and composition of trees: In the newer suburbs on the west side of the city, the trees were often larger and more likely to be native compared to those on the east side, which had more deciduous, nonnative species (Oke et al.). This distribution is still evident today, although more diverse street tree plantings have begun to replace planted monocultures.

Some of the original vegetation of the area is retained in Greater Vancouver's large park system, and this resembles the dense coniferous forest of the coastal western hemlock (CWH) zone, with its shrub-dominated understory. The climax vegetation of this zone is generally dominated by a canopy of western red cedar (*Thuja plicata*) and western hemlock, with Douglas fir in drier areas and smaller numbers of Sitka spruce (*Picea sitchensis*), yellow cedar (*Chamaecyparis nootkatensis*), and lodgepole pine (*Pinus contorta*). Many species of nonnative vegetation, such as English holly (*Ilex aquifolium*) and Himalayan blackberry (*Rubus discolor*) are now also common in many Greater Vancouver parks and urban areas.

### Bird Surveys

I collected relative abundance data for individual bird species at 285 point-count locations along four roadside transects in Greater Vancouver (see Figure 1; Melles, 2001). Point-count locations (with an interstation distance of 250 meters) were sampled once each year during the breeding season, 24 June–13 July 1997 and 1 May–1 June 1998, to maximize the number of sites that could be surveyed over the landscape. Birds flying over the point-count stations were not recorded because they were considered unlikely to be breeding in the area. The fixed-radius methodology (50 meters) was followed, and birds were recorded for a period of five minutes (see DeGraaf, Geis & Healy, 1991; Ralph, Geupel, Pyle, Martin & DeSante, 1993). All bird surveys were conducted on clear days during the first four hours following sunrise, to coincide with peak singing activity. I combined the bird-community data for 1997 and 1998 by selecting the maximum abundance for the two years at each point-count station. These abundance data were then averaged over all point-count stations within a given neighborhood census-tract area. Although the maximum value may be an optimistic estimate, it is likely to be a more accurate estimate of abundance at a particular site than the mean of one survey in each of two years (see Vander Haegen, Dobler & Pierce, 2000).

Because point-count data were only collected once during the breeding season of each year, 1997 and 1998, and the timing of data collection coincided with spring migration in 1998, this may have biased the abundance data in 1998. Some of the birds counted may have been migrants passing through the area. The focus in this study was on resident species of birds; thus, only species found in more than 10% of the census-tract neighborhoods and known to breed in the area were included in the analyses (n = 23 species). Though it is still possible that some of the individual birds recorded in 1998 were migrants, there were no significant year effects found between the community of birds recorded in 1997 and 1998 (data not shown, Melles, 2001).

### Socioeconomic Data

According to Statistics Canada census definitions, a census family refers to a married or common-law couple (with or without children) or a lone parent of any marital status (Statistics Canada, 1996). I selected mean family income and the number of people holding a university degree (bachelor's or higher) from the 1996 Statistics Canada census as a measure of socioeconomic status. Of course, a number of economic variables could have been selected, but many were highly correlated, and these two variables should capture a large amount of the variability in both the income of an average residence and the education of an average person living in such a residence. I estimated neighborhood population density, spatial segregation among social classes, and racial composition using the census-tract data. All socioeconomic census data represented 20% of the total census population, whereas population estimates were absolute numbers of people.

I used the number of houses constructed during different time periods to estimate time since neighborhood development. This type of variable has been used before in a related study (Martin et al., 2004) to estimate the "time since disturbance," and it assumes that prior to the development, the area would have retained some of the original native vegetation. Logging operations may have cleared some of the land prior to neighborhood development, but the disturbance caused by construction of houses and impervious roadways removes land from natural regenerative processes. Thus, it is reasonable to estimate time since disturbance by the number of houses constructed in a census neighborhood during a given period of time.

### Data Analysis

To characterize the relationships between socioeconomic status, time since disturbance, spatial segregation among ethnic backgrounds, and the abundance of different bird species one might expect to see in a given neighborhood, I used redundancy analysis (RDA) (ter Braak & Šmilauer, 1998; Legendre & Legendre, 1998). RDA is akin to direct gradient or regression analysis done in multivariate species space (ter Braak & Šmilauer). RDA relates abundance data from a species matrix to a matrix of environmental data using multiple linear regression techniques, and it assumes that species have linear responses to ecological gradients. That is, the abundance of a species is expected to increase linearly along a gradient. Linear responses may arise when species distributions extend beyond the extremities of the gradient sampled (Austin, 2002).

I selected redundancy analysis as opposed to canonical correspondence analysis (CCA) because a preliminary examination of the data showed that the length of the species gradient was short with respect to the socioeconomic variables (gradient length = 1.91), and RDA is recommended when gradients are short (i.e., < 3 SD; ter Braak & Šmilauer, 1998). The species matrix consisted of the average abundance per census-tract area of 23 species of birds (Table 2). Although 48 species were detected in the study area, only birds present in more than 10% of the 44 census neighborhoods were included in the analysis in order to exclude potential migrants that were not breeding in the area (as noted above). The final species matrix was related to a linear combination of 12 socioeconomic variables (Table 3). The significance of the RDA ordination of species and human socioeconomic relationships was investigated by performing a randomization test on the projected relationships (ter Braak & Šmilauer); 199 random permutations were performed on the significance of the ordination axes. Randomizations were spatially restricted by the linear transects in order to ensure that shuffling was not entirely random across the study area. Data were not sampled randomly across the study area, and hence permutations were restricted within the line transects (CANOCO 4; ter Braak & Šmilauer). Redundancy analysis allows one to infer which variables best explain the variation in species distributions because the most important variables load highest on the first axis.

### Variance Partitioning

Census tracts in Greater Vancouver did not cover equal amounts of area (Figure 1), and this could result in some census neighborhoods having more species simply because the area sampled was larger. Variance partitioning is a technique used to examine the relative contribution of different factors while controlling for covariables that may have overlapping effects (Bocard, Legendre & Drapeau, 1992). I was interested in partitioning out the amount of variation in the species data that could be attributed to the area of a census neighborhood. In addition, I wanted to determine the relative contribution of another potentially confounding factor—the spatial structure of the species data (Bocard et al., 1992). As suggested by Bocard and colleagues, spatial structure was modeled using trend surface analysis. I used the spatial coordinates of the census-tract centroids, defined using the universal transverse mercator, North American datum 83 projection, as covariables in partial RDA analyses. Only the X and Y coordinates were used because all higher-order terms (for example, $X^2$, XY) were highly correlated with these two. I used variance partitioning to partition out the amount of variation in the species data that could be attributed to socioeconomic variables, spatial variables, and area (Bocard et al., 1992; Cushman & McGarigal, 2002).

## Results

A map of Greater Vancouver depicting the study area (Figure 1) shows neighborhoods (delineated as census tracts outlined in black) in relation to their proximity to large urban parks. Figure 2 (a–c) shows that census-tract neighborhoods with the lowest mean family incomes (2a) are also the areas with the highest proportion of people of aboriginal ethnicity (i.e., North American Indian, Métis, or Inuit and/or those who reported being a Treaty Indian or a Registered Indian as defined by the Indian Act of Canada) (2b). These neighborhoods also have the highest population densities in Greater Vancouver (2c), have the fewest small parks, and are the farthest away from large urban parks (Figures 1 and 2).

Mean family income had the strongest positive correlation with RDA axis I (Figure 3; Table 3, interset correlation with RDA I = 0.47), indicating that this variable has a strong influence on the separation of the bird species data along a socioeconomic gradient; the majority of native bird species were positively related to increasing socioeconomic status. Mean family income was followed closely by census-tract area (Figure 3; Table 3, interset correlation with RDA I = 0.43). I expected that area would be a strong explanatory variable given that the species-area relationship is one of the most general patterns found in ecology. In short, as the logarithm of the area sampled increases, more and more species are detected (Pileou, 1966).

The positive relationship between the number of people holding university bachelor's degrees (or higher) and native avian species diversity (Figure 3; Table 3, interset correlation with RDA I = 0.30) likely reflects the location of Vancouver's university campuses. Both the University of British Columbia and Simon Fraser University are surrounded by park space and university endowment lands. So the diversity of birds in these areas reflects park habitat rather than people's personal preferences or advanced learning shaping the local habitat of their neighborhood.

Two other interesting findings can be ascertained from the RDA ordination (Figure 3). First, there appears to be no discernable relationship between time since disturbance and the distribution of bird species, contrary to expectations. Although several of the disturbance variables were significantly correlated with the RDA axes (Table 3), there is no apparent trend through time—the community of birds in neighborhoods built primarily prior to 1946 is similar to the community of birds near newer houses built between 1990 and 1996. But older neighborhoods have more nonnative species of birds (Figure 3). The second noteworthy finding is that persons with aboriginal ethnicity reside predominately in areas with fewer bird species, and the bird species that do occur in these areas are generally nonnative. This finding further describes the spatial segregation indicated in Figure 1.

Variance partitioning was used to determine how much of the variation in the distribution of bird species can be attributed to either the area of the census tract surveyed or the tendency for bird species distributions to have some level of spatial structuring. Figure 4 shows that the majority of the variance in the urban bird community of Greater Vancouver can be explained by socioeconomic and time since disturbance variables (neighborhood age was grouped with the social variables). Almost 50% of the variation in bird abundance data can be attributed to all three factors (social, area, and space) combined. However, social variables account for 29.8% of that variation; census area accounts for 6.1%; and spatial structuring in the bird community accounts for almost 7.8%. Only 6.1% of the variation in the avian community is shared between socioeconomic variables and either spatial variables or area. Census area (hectares) and the two spatial variables did not share any amount of variation. Although social variables and time since disturbance were able to explain more variance, we should be cautious about interpreting this to mean that these variables are better or the more important factors in the system. That's because there were four times more social and disturbance variables than area and spatial variables combined (see Table 3).

## Discussion

According to Turner, Nakamura, and Dinetti (2004), most of the world's human population lives in biological poverty. In one study examining the relationship between human population density and species diversity at global scales, they examined data from five cities around the world and found an inverse relationship between the numbers of humans and the diversity of birds in neighborhoods (scaled at one square kilometer). My work in Greater Vancouver substantiates their findings from Berlin, Germany; Washington, D.C., USA; Florence, Italy; Chiba, Japan; and Tucson, Arizona, USA. (Only 6 of 23 species of birds in Greater Vancouver were related to increasing numbers of humans; see left-hand side of Figure 3). But the bird-human relationship is not necessarily straightforward in Greater Vancouver: Complex socioeconomic and cultural factors are also correlated with the diversity of birds. In my study, neighborhoods of higher socioeconomic status tended to have more native species of birds than ones of lower socioeconomic status (i.e., those predominantly composed of aboriginal peoples). Furthermore, Turner, Nakamura, and Dinetti (2004) argue that human ability to assess the overall ecological health of an area diminishes as new generations are exposed to poor ecological conditions. If this is indeed the case, then certain city neighborhoods with persistent low socioeconomic status are in danger of becoming self-perpetuating and self-segregating areas of low biodiversity.

It is perhaps not surprising that socioeconomic status was strongly correlated with the community of birds in Greater Vancouver (Figure 3). If socioeconomic status limits a family's ability to purchase a house in the neighborhood of a large park, then this indeed follows from that expectation. Melles, Glenn, and Martin (2003) showed that park area and coniferous and deciduous tree cover (within one kilometer of avian point-count stations) were significantly related to the likelihood of observing most native species of birds. Even the likelihood of finding more common species like the American robin (*Turdus migratorius*) significantly increased with the amount of park area in the vicinity. Greater Vancouver is relatively unique (and fortunate) in having large remnants of historic vegetation; however, the unequal spatial distribution of these remnants constrains access to these large parks. Land values in urban areas are often heavily influenced by the proximity of parks and other green space. Affordable urban housing is often limited to abandoned commercial and industrialized areas of the city, where there is more impervious surface and less green space.

I did not find a relationship between time since disturbance and the bird community, though I expected that older neighborhoods would have more well-advanced vegetation (larger trees and shrubs) and thus higher bird species richness and abundance. Upon closer inspection of the data, it became clear that the amount and type of vegetation in older areas could not be generalized. One of the oldest neighborhoods in Vancouver is located close to the downtown core, just to the east of Stanley Park (Figure 1, labeled "Historic A"). Because this area is so close to the urban center, the density of buildings is high, and much of the vegetation has been removed, leaving little habitat for birds. Another historic neighborhood in Vancouver is located in the west end (Figure 1, labeled "Historic B"). Here, many large mature street trees—native species such as Douglas fir and western hemlock—remain standing today and provided habitat for a variety of bird species. Several neighborhoods to the east of the downtown core, developed primarily between 1946 and 1960 (data not mapped), were initially planted with many nonnative cherry and plum street trees. These ornamental fruit trees have short life spans and require replacement, and as such, these plantings have not been conducive to maintaining a diverse bird community. Therefore, contrary to my expectations, older neighborhoods did not necessarily have a more well-advanced vegetation community in Greater Vancouver.

Landscape-scale spatial heterogeneity in urban areas is established and maintained in a "top-down" way by formal institutions like city planning departments (through zoning bylaws), public works, and courts (Grimm et al., 2001). In Vancouver, there are historic reasons for the spatial distribution of parks (for example, federal endowments of land to the universities and the historic transfer of lands previously held in federal government reserves, such as for Stanley Park; see Figure 1). However, less formal "bottom-up" actions such as tree plantings, community gardens, and park maintenance by community groups, families, and associations can have a profound influence at the local level and also contribute to large-scale spatial heterogeneity. Residential and local community gardens planted with large berry-producing shrubs (e.g., salmonberry, blackberry, and elderberry) and conifer trees, for instance, had a higher likelihood of being occupied by many species of birds in Greater Vancouver, including bushtits (*Psaltriparus minimus*) and spotted towhees (*Pipilo maculatus*) (Melles et al., 2003). It must be noted, however, that impoverished areas tend to have lower levels of residential involvement in neighborhood tree planting and community efforts, and this could reinforce social and spatial segregation (Pedlowski et al., 2002).

Social scientists have long examined how human perception, choice, and action drive the political, economic, and cultural decisions that lead to—and respond to—changes in urban areas (Jacobs, 1961; Grimm et al., 2001). Why is there such extreme evidence of ethnic segregation in Greater Vancouver, particularly with respect to the aboriginal population? And what does the unequal distribution of environmental amenities mean in terms of social justice? These two questions reflect some of the most complex issues affecting contemporary urban life (Pedlowski et al., 2002). Jacobs (1961) argues that some of the factors that foster healthy and ethnically diverse neighborhoods in urban areas are community involvement, low rent-to-income levels, cultural and spatial heterogeneity, and active social interactions at the street level. In Greater Vancouver, there are a variety of other historic and social factors that also influence the extreme patterns we see. In order to turn these patterns around, we need to see far greater social, economic, scientific, and community involvement in neighborhoods that are often ignored or avoided.

Understanding why certain neighborhoods are ignored by city planners and avoided by urban residents is the first step to changing these patterns. Ethnic and economic segregation exists in many cities worldwide, but in Canada it is especially evident in Vancouver. It is a complex and systemic problem involving many factors such as intravenous drug and alcohol abuse, poverty, homelessness, and neglect. Many of these problems are most rampant among aboriginal people (Riley, 1998). The following statistic demonstrates just how bad the situation can get: Vancouver's east side set the world record in 1998 for the largest increase in the number of HIV cases among intravenous drug users (Riley, 1998; Nolin, 1999).

Often there is a social stigma associated with poverty and with people who are addicted to drugs and alcohol, as though they have brought their circumstances and illnesses upon themselves (Nolin) and as though their position bears no reflection on society as a whole. This sort of social stigma perpetuates ethnic spatial segregation. Many people refuse to admit that drug consumption is a health problem (Nolin), even as the costs for Canada (in 1992) were estimated at more than $18 billion (Riley), including the combined costs of drug enforcement, loss of productivity due to morbidity, premature deaths, and direct health costs. Recent initiatives under the city of Vancouver's draft plan "for the prevention of harm from psychoactive substance abuse" (2005) hold considerable promise for turning these patterns around. This plan aims to prevent harm by approaching the problem from the perspective of population health, adequate housing and employment, access to health care, and individual and community-based approaches.

Interestingly, approaches designed to prevent harm from substance abuse are the same sort of top-down and bottom-up approaches that could prevent the further deterioration of urban ecological health. For example, creating healthy school environments (including those of schoolyards) and supporting parents to help engage youth in community and social initiatives like urban renewal should have positive ecological and health outcomes. Research has shown that concern for natural features can be fostered by direct interaction with such features (Turner et al., 2004; Bixler, Floyd & Hammitt, 2002; Schultz et al., 2005). Ultimately, social health may be inextricably linked to ecological health.

People living in the poorest neighborhoods, at the extreme end of the socioeconomic gradient, have lower levels of residential involvement in neighborhood tree planting and community green-up efforts than better-off citizens, probably because they are more concerned with the immediate issues of day-to-day survival. Although biological poverty in these neighborhoods does have implications for environmental justice, it is unlikely, in the near term, that fewer species of birds in these areas will lead to reduced support for the protection of biological diversity in Vancouver: The numbers of people in this very vulnerable segment of society are relatively small here (City of Vancouver, 2005).

However, as more and more people in middle- to low-income neighborhoods grow up surrounded by a depauperate bird community, the ability to assess ecological health may indeed diminish. Urban citizens perceive and appreciate the "nature" they know, so as new generations are exposed to poor ecological conditions, the dissociation from biological diversity could lead to lower popular support for natural diversity at regional and even national levels. With increasing population growth in urban neighborhoods, it is becoming more important to plan for the kinds of wildlife values we wish to uphold at municipal, provincial or state, and national levels. Ad hoc urban planning, with no attention paid to the importance of vegetation and native wildlife for overall community health, will lead to environmental erosion. Though several community green-up initiatives do stem from bottom-up groups, these are unlikely to be able to deal with the magnitude of the problem. It is reasonable to conclude that minimum standards and bylaws need to be set for such amenities as native tree and shrub plantings and removals, as well as the maintenance of drought-resistant residential garden plants.

In terms of scientific involvement, there are a number of logistical and social obstacles associated with doing research in urban habitats, and this is perhaps why experimental studies are so scarce (Table 1). First, gaining access to numerous private properties can be a daunting task for scientific researchers, especially younger ones, who are perhaps not comfortable communicating and justifying their research to the public. Second, there are issues of privacy and safety: Some people might not want a researcher peering with binoculars into their backyard. Third, there is inadequate financial support and political backing for the study of urban areas, possibly because of the attitude that there are more deserving areas for our conservation and research dollars. Fourth, urban areas are extremely altered, complex systems that provide coincidental habitat for wildlife in an environment designed primarily to provide a variety of socioeconomic services that satisfy human needs, preferences, and desires. Wildlife species in urban areas are viewed as incidental (Clergeau et al., 2001) and perhaps even nonessential inhabitants. Birds may indeed be a source of pleasure for people in urban, suburban, and rural areas alike (Clergeau et al., 2001), but in urban areas they often go unnoticed.

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## Glossary

**Canonical correspondence analysis (CCA):** A standard method for multivariate direct gradient analysis (regression), whereby multiple species abundance data are related to linear combinations of measured environmental variables (ter Braak, 1986). This method assumes that species abundances vary along environmental gradients in unimodal ways.

**Centroid:** The center of gravity for a homogeneous area.

**Fixed-radius methodology:** In this methodology, only birds observed within 50 meters of a point-count circle's center are used to estimate relative abundances.

**Maximum abundance:** The maximum species abundance observed over all the years of a study (e.g., 1997 or 1998 for this study), recorded for each point-count station.

**Mensurative:** Adapted for measuring.

**Point-count stations:** A stop location along a bird survey transect where an observer records all birds heard or seen within a given radius of the stop for a set period of time (e.g., three to five minutes).

**Randomization test:** A test that uses randomly generated numbers for statistical inference (see http://ordination.okstate.edu/permute.htm).

**Redundancy analysis (RDA):** Another standard method for direct gradient analysis done in multivariate space. Multiple species abundance distributions are assumed to have linear relationships to environmental gradients (i.e., linear species response curves).

**Regression analysis:** Any statistical method in which the mean of one or more random variables is predicted conditioned on other (measured) random variables (see http://en.wikipedia.org/wiki/Regression_analysis).

**Shuffling:** A method of randomly rearranging data. Data can be shuffled in various ways for randomization tests (e.g., across all point-count stations surveyed or shuffled among point-count stations within transects, but not between them).

**Transect:** A line used in ecological surveys to provide a means of measuring and representing graphically the distribution of organisms (*Oxford Dictionary of Ecology*).

**Trend surface analysis:** An analysis technique designed to separate observed data into large-scale (spatial or regional) components and residual components.

**Unimodal:** Pertaining to a distribution of data with only one mode, or peak, such as a standard bell curve.

**Universal transverse mercator:** A map developed by the National Imagery and Mapping Agency (NIMA) using a noncurved rectangular grid system. In this grid, the world is divided into 60 north-south zones, each covering a strip 6° wide in longitude (see http://www.uwgb.edu/dutchs/FieldMethods/UTMSystem.htm).

**Year effects:** Significant year-to-year differences (in terms of species richness and abundance) in the community observed.

**Figure 1. Study area, parks (green areas), census tracts (black outlines), and transect locations (four altogether, represented by purple, yellow, red, or black dots) for breeding bird and socioeconomic data in the Greater Vancouver area, British Columbia, Canada.**

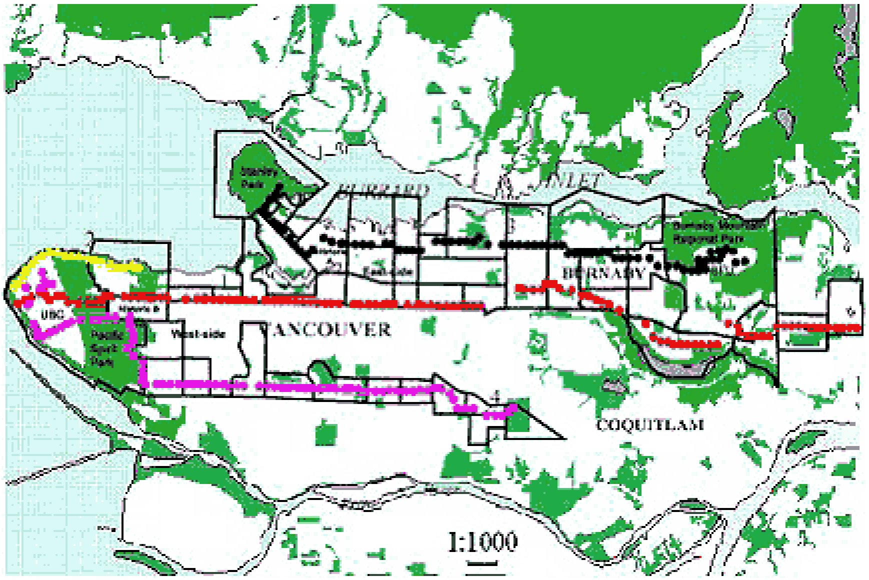


**Figure 2a–c. Census-tract neighborhoods in Greater Vancouver depicting a) trends in mean family income levels (20% of sample data), b) aboriginal population densities in 1996, and c) total population densities in 1996.**

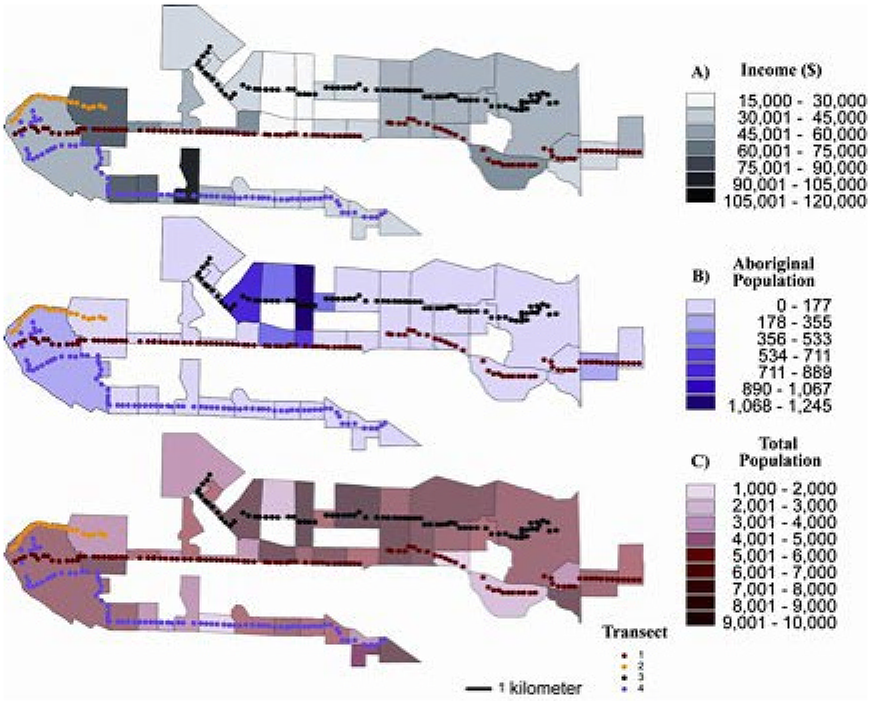


**Figure 3. Redundancy analysis, ordination bi-plot examining the strength of association between human social variables and 23 bird species in an urban bird community, Greater Vancouver, BC. (See Table 2 for bird name abbreviations.) Species and socioeconomic variables are represented by arrows that indicate the direction in which the variables are increasing.**

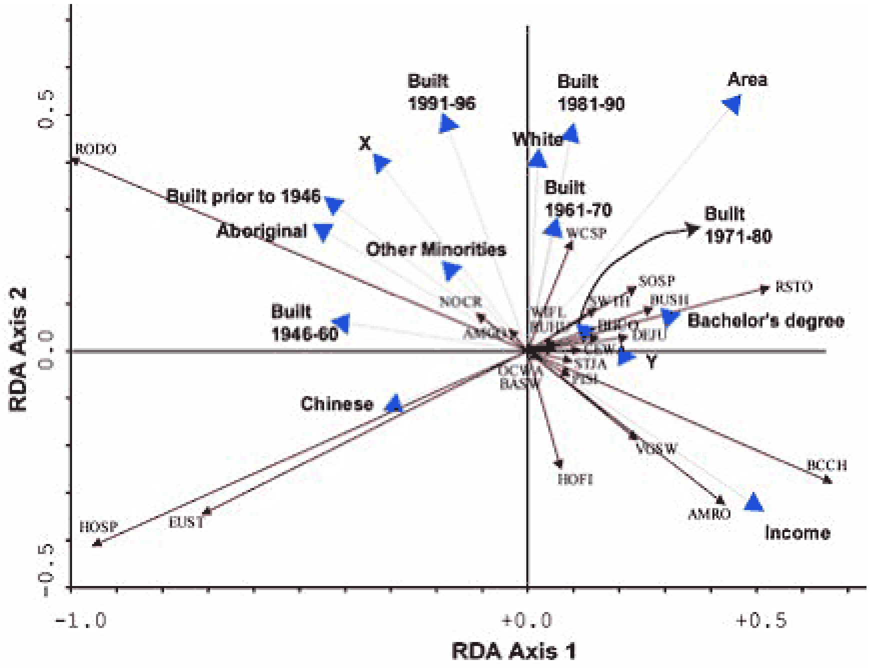
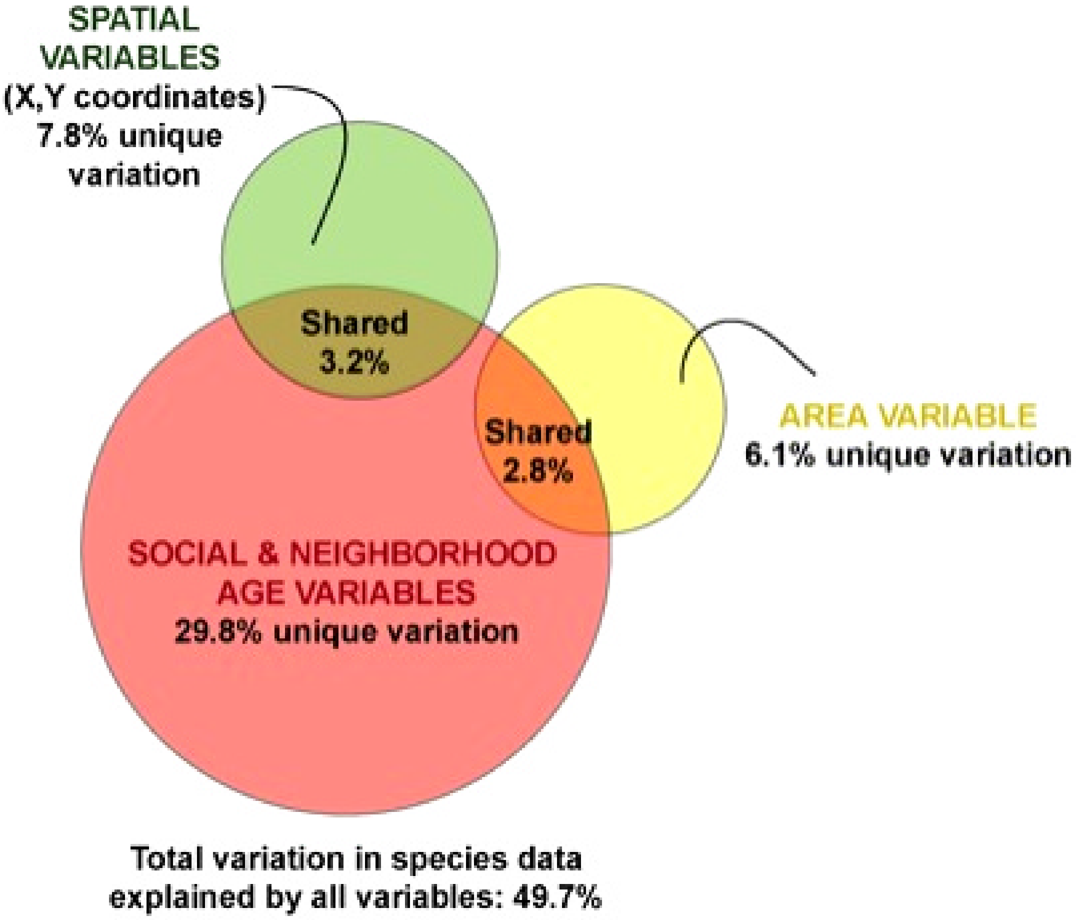


**Figure 4. Variance decomposition based on partial redundancy analyses of the bird community in Greater Vancouver, BC**



Pink represents variance explained by social and neighborhood-age variables, after removing the effects of covariates area and space (XY coordinates).

Yellow represents variance explained by census-tract area alone, after removing the effects of the space covariate and social + neighborhood-age variables.

Green represents variance explained by space alone, after removing the effects of the area covariate and social + neighborhood-age variables.

Amounts of shared variance were calculated by running a further series of partial redundancy analyses and calculating the joint or shared effects of 1) social + neighborhood-age variables, constrained by area, 2) social + neighborhood-age variables constrained by space, and 3) area, constrained by space.

**Table 1. Future areas of research on urban birds: key research questions.**

| **Research Area** | | **References**[1] |
|---|---|---|
| DEVELOPMENT PLANNING | Effects of different human settlement patterns (especially in the tropics) on bird diversity, abundance, and productivity. Where and what styles of development will have the least impact? | Marzluff et al., 2001. |
| LANDSCAPE CONTEXT | What is affect of surrounding neighborhood tree cover (for instance) on bird species richness, abundance, and nest productivity? | Donnelly & Marzluff, 2004, as well as some of the multi-scale studies. |
| MULTI-SCALE | What is the relationship between scale of investigation and the strength of species habitat relationships in urban areas? Does this relationship change if you move between cities or between land uses (e.g., forested areas)? | Several large scale, investigations over a short duration of time (e.g., Hostetler, 2001; Fernández-Juricic, 2002; Jokimaki, 2003; Melles et al., 2003). |
| DISPERSAL | Preferred dispersal pathways in an urban setting? Do birds tend to use urban corridors such as street trees and/or back yard lanes? Are birds willing to cross larger openings in urban areas than in areas with more continuous cover? What is mortality risk of crossing different land cover types in urban areas? | No studies known in urban areas, but see Desrochers & Hannon, 1997; St. Clair et al., 1998, for forest fragmentation examples. |
| TOP-DOWN | Is aerial predation pressure lower in urban areas? What about brood parasitism rates? | Bolger et al., 2001; Blair, 2004. |
| BOTTOM-UP | Food availability? | Bolger et al., 2001 |
| LONG-TERM POPULATION DYNAMICS | What are the long-term population dynamics of bird species in urban areas? Are they unique in terms of disease, contaminant loadings, evolutionary pressures, and/or predation risk? | See Grimm et al., 2001, for initiation of LTER studies. |
| RICHNESS THRESHOLD | Are there thresholds in the relationship between bird diversity and the amount of impervious surface or native land cover? Is there a threshold level of connectivity necessary to maintain diversity or meta-population structure (e.g., street tree corridors, urban park networks)? | No studies known in urban areas, but see Fahrig, 2002, for theoretical predictions. |
| META-POPULATION STUDIES | Do sensitive species that breed in fragmented urban parks (e.g., shrub nesters) demonstrate meta-population structure? | No studies known, but see for example, Brooker & Brooker, 2001. |
| OPTIMAL FORAGING | Do birds in urban areas have different optimal foraging strategies? | Shochat, 2004 |
| FECUNDITY AND MORTALITY | How does mortality vary in different life stages (e.g., nest predation, juvenile mortality, and adult mortality)? How does fecundity vary by species and/or within species in different cities (e.g., by latitude/longitude?) | Not much work has been done, but see Lepczyk et al., 2003 for cat predation risk. |
| COMPLEX SYSTEMS: PROCESS-BASED | What are the ecological processes and interactions involved in maintaining an urban bird community (i.e., food availability, extinction | No studies known |

| | | |
|---|---|---|
| ECOLOGY | dynamics, source/sink dynamics, predation, parasitism, disease, dispersal, adaptation)? | |
| INVASIVE SPECIES | What are features or conditions for successful establishment of non-native species in cities around the world? Why are some invasive species successful and not others (e.g., Eurasian tree sparrow or crested myna in North America)? | Blair, 2001 |
| ANIMAL BEHAVIOUR | Do birds in urban areas tend to be more socially gregarious breeders? What are the dynamics of inter- and intra-specific competition in urban areas? | McGowan, 2001 |
| SOCIOECONOMICS (A) | How do human socioeconomic factors influence perception of urban bird species? How do urban surroundings influence environmental preferences about bird species? | Clergeau et al., 2001;<br>Kaplan & Talbot, 1988 |
| SOCIOECONOMICS (B) | What are key human socioeconomic correlates with urban bird diversity and abundance? What are the mechanisms involved in creation of spatial segregation of wealth levels, bird diversity, and ethnicity? | Turner et al. 2004 |
| SOCIOECONOMICS (C) | What is the best way to impact bottom-up community-level initiatives (e.g., tree plantings, community gardens)? How do top-down planning initiatives influence urban birds? | No studies known |

[1] Not intended to be a comprehensive list of all research but to serve as an example of research done in the area (and/or a call for research).

**Table 2. Number of census tracts where 23 species of birds were recorded at point-count stations in the Greater Vancouver area (1997–1998). Only species that were found in more than 10% of the census tract areas were included in the analysis. Bird names follow Campbell (1998). Italics and boldface indicate nonnative species.**

| **Species Code** | **Common Name** | ***Scientific name*** | **Presence (44 Census Tracts)** | **Max relative abundance on occupied point counts** |
|---|---|---|---|---|
| *EUST* | European Starling | *Sturnus vulgaris* | 44 | 4.17 |
| NOCR | Northwestern Crow | *Corvus caurinus* | 43 | 2.10 |
| *HOSP* | House Sparrow | *Passer domesticus* | 43 | 4.41 |
| HOFI | House Finch | *Carpodacus mexicanus* | 42 | 2.20 |
| AMRO | American Robin | *Turdus migratorius* | 42 | 1.72 |
| BCCH | Black-capped Chickadee | *Poecile atricapillus* | 41 | 1.99 |
| *RODO* | Rock Dove | *Columba livia* | 35 | 3.24 |
| VGSW | Violet-green Swallow | *Tachycineta thalassina* | 31 | 2.54 |
| BUSH | Bushtit | *Psaltriparus minimus* | 21 | 3.21 |
| SPTO | Spotted Towhee | *Pipilo maculatus* | 18 | 1.62 |
| WCSP | White-crowned Sparrow | *Zonotrichia leucophrys* | 17 | 1.34 |
| BASW | Barn Swallow | *Hirundo rustica* | 15 | 2.32 |
| DEJU | Dark-eyed Junco | *Junco hyemalis* | 12 | 1.19 |
| SOSP | Song Sparrow | *Melospiza melodia* | 10 | 1.68 |
| STJA | Steller's Jay | *Cyanocitta stelleri* | 10 | 1.17 |
| BHCO | Brown-headed Cowbird | *Molothrus ater* | 9 | 1.42 |
| AMGO | American Goldfinch | *Carduelis tristis* | 9 | 1.24 |
| CEWA | Cedar Waxwing | *Bombycilla cedrorum* | 9 | 1.14 |
| OCWA | Orange-crowned Warbler | *Vermivora celata* | 8 | 1.20 |
| PISI | Pine Siskin | *Carduelis pinus* | 7 | 1.26 |
| SWTH | Swainson's Thrush | *Catharus ustalutus* | 5 | 1.39 |
| RUHU | Rufous Hummingbird | *Selasphorus rufus* | 5 | 1.17 |
| WIFL | Willow Flycatcher | *Empidonax trailii* | 5 | 1.00 |

**Table 3. Interset, linear correlation coefficients between socioeconomic variables of census tract areas and the first and second redundancy analysis axes that describe the bird community in Greater Vancouver, British Columbia.**

| **Variable code** | **Description** | **Correlation of socioeconomic variables with** | |
|---|---|---|---|
| | | *Axis I* | *Axis II* |
| Area (ha) | Census tract area (hectares) | 0.43* | 0.43* |
| X | UTM easting (m) NAD 83 | 0.22 | -0.01 |
| Y | UTM northing (m) NAD83 | -0.32* | 0.34* |
| Income | Mean family income (20% data) | 0.47* | -0.26 |
| Bachelor's degree | With bachelor's degree or higher (20% data) | 0.30* | 0.06 |
| Built prior to 1946 | Private dwelling – period of construction prior to 1946 (20% data) | -0.40* | 0.25 |
| Built 1946-60 | Private dwelling – period of construction between 1946 and 1960 (20% data) | -0.39* | 0.05 |
| Built 1961-70 | Private dwelling – period of construction between 1961 and 1970 (20% data) | 0.06 | 0.22 |
| Built 1971-1980 | Private dwelling – period of construction between 1971 and 1980 (20% data) | 0.14 | 0.04 |
| Built 1981-90 | Private dwelling – period of construction between 1981 and 1990 (20% data) | 0.09 | 0.38* |
| Built 1991-96 | Private dwelling – period of construction between 1991 and 1996 (20% data) | -0.17 | 0.39* |
| Aboriginal | Population native aboriginal (20% data) | -0.42* | 0.21 |
| White | Population non-visible minority | 0.02 | 0.35* |
| Other Minority | Population of all other visible minority groups (20% data) | -0.17 | 0.15 |
| Chinese | Population chinese (20% data) | -0.29 | -0.10 |

* $p<0.05$

# Microhabitat Selection and Singing Behavior Patterns of Male House Finches (*Carpodacus mexicanus*) in Urban Parks in a Heavily Urbanized Landscape in the Western U.S.*

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## Abstract

We assessed the role of park size, habitat structure, human disturbance (pedestrian rate and ambient noise), and the number of conspecifics in the distribution, spacing, and singing behavior of male house finches (*Carpodacus mexicanus*) in urban parks in southern Los Angeles County and north Orange County, California. We found that the probability of house finch males occupying urban parks increased with park size and tree structure (total tree cover, tree height, and the number of stems 30 to 50 centimeters in diameter)—two features that may increase the availability of suitable nesting substrates. Nearest neighbor distance between singing males increased with denser vegetation structure (e.g., number of stems), probably because of better nesting and foraging resources, or greater availability of protective cover, which would reduce aggregation. Males increased their singing rates in the most exposed parts of their perches (upper and outer portions). They also raised the low frequency of their songs to reduce the masking effects of high ambient noise levels. However, the number of notes per song decreased with ambient noise, and since females are attracted to long songs, this could decrease mating opportunities. Our results point out some of the mechanisms house finch males use to increase their breeding success in urbanized areas and suggest that this success may vary depending on the specific spatial location of nesting areas within a city.

**Keywords:** ambient noise, birds, distribution, males, spacing behavior, singing rate, songs, urban ecology, urban parks

## Introduction

Urban sprawl has modified natural landscapes by changing the availability and configuration of suitable habitat for wildlife and by altering the nature and frequency of human-wildlife interactions (Fernández-Juricic & Jokimäki, 2001; Marzluff, 2001). The size of remnant habitat fragments is a primary influence on the number of birds that urban

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habitats can sustain (Fernández-Juricic, 2000; Crooks, Suarez & Bolger, 2004), mainly because of the area requirements of interior, rare, and low-abundant bird species (Crooks, Suarez, Bolger & Soulé, 2001; Fernández-Juricic, 2002). Suitable fragments might be urban parks or natural areas surrounded by urban development. Vegetation structure has also been found to increase species richness by enhancing key habitat resources that facilitate the colonization of certain species (Donnelly & Marzluff, 2004; Feldman & Krannitz, 2004; White, Antos, Fitzsimons & Palmer, 2005).

High levels of human visitation to these urban fragments may reduce the spatial and temporal access certain birds have to suitable resources, and decrease the chances of park occupation by low-disturbance-tolerant species (Fernández-Juricic, 2002). Furthermore, human-generated ambient noise (car traffic, trains, airplanes, industries, etc.) can mask the communication systems of some birds (Rabin & Greene, 2002; but see Leader, Wright & Yom-Yov, 2005), causing them to adjust the vocal structure of their songs (Slabbekoorn & Peet, 2003; Brumm, 2004). Taken together, this evidence suggests that urban habitats are complex environments, and that different species use various habitat-selection mechanisms for breeding purposes (Sedlacek, Fuchs & Exnerova, 2004).

So far, researchers have mainly concerned themselves with minimizing the decline of interior bird species in urban landscapes, because these species appear to be more sensitive to urbanization (Savard, Clergeau & Mennechez, 2000; Fernández-Juricic & Jokimäki, 2001; Marzluff & Ewing, 2001; Chace & Walsh, in press). However, edge species, many of which have thrived in urban areas around the world (Marzluff, 2001), make interesting models for understanding the life-history traits and ecological factors that enable some birds to adapt successfully to human-dominated landscapes. We studied one such species, the house finch (*Carpodacus mexicanus*), which is considered native to the western U.S. but has spread throughout the eastern portion of North America. It inhabits open and semiopen areas, particularly in urban and suburban locations (Hill, 1993, 2002).

Our goal was to assess the role of fragment size, habitat structure, human disturbance (pedestrian rate and ambient noise), and number of conspecifics in the distribution, spacing, and singing behavior of male house finches in urban parks. We focused on the birds during the breeding season to understand the potential mechanisms influencing mating opportunities. Male house finches display bright carotenoid-based plumage coloration (Hill, 2002) and emit a song comprised of a series of notes, sometimes followed by a trill and a buzz (Bitterbaum & Baptista, 1979; Hill 1993). Male songs appear to be more involved in female attraction than in male-male competition due to the apparent lack of territoriality in this species (Thompson, 1960; Hill, 1993, 2002; Nolan & Hill, 2004). However, little is known about the effects of male competition on spacing and singing behavior of house finches (Hill, 1993).

Specifically, we studied (a) the effects of park size, vegetation structure, and pedestrian rate on the occurrence of male house finches in urban parks; (b) the influence of the number of singing males and tree structure (as indicators of competition and the availability of suitable vegetation) on the spacing behavior of males; (c) the effect of the number of males and degree of exposure in perches on their singing rate (number of songs per unit time); and (d) the role of ambient noise on the frequency and duration of male songs.

## Methods

### Study Area

We studied the house finch (*Carpodacus mexicanus*) in its native California, where no geographic variations in vocalization are reported to exist (Bitterbaum & Baptista, 1979). (In other parts of the house finch's distributional range, different dialects have been recorded; see Pytte, 1997, and Tracy & Baker, 1999).

The study was conducted at several sites in south Los Angeles County and north Orange County during the spring of 2005, mostly in the city of Long Beach. We chose 35 parks (Table 1) that were representative of the variability in size of the parks in the region. All had wooded areas with tree cover, introduced and native shrub species, and areas of watered grass.

### Male House Finch Surveys

Each park was surveyed two or three times during the spring of 2005, on weekday mornings from 6:00 a.m. to 9:30 a.m. We recorded the presence and number of male house finches and surveyed male singing and courtship behavior. In parks greater than two hectares in size (23 of the 35 parks), we set up 100-by-50-meter transects in both the interior and at the edges of parks, each separated by 100 to 200 meters (Järvinen & Väisänen, 1977). Prior to beginning the surveys, observers were trained to visually estimate 25 meters on each side of the transect central path with less than 10% error. The same transects were used in each visit. The number of transects per wooded park was established in proportion with a logarithmic scale of the size of each park. In each of the parks less than two hectares in size (12 parks), we sampled the whole park area for a period of time that was proportional to the time used in sampling the line transects (see more details of the survey techniques in Fernández-Juricic, 2000, 2004).

Three independent factors were considered: park size, vegetation structure, and human disturbance. Park sizes were derived from the city websites of Long Beach and Seal Beach.

We measured vegetation-structure traits in 25-meter-radius circular plots distributed at 30-meter intervals along transects in parks larger than two hectares, and randomly in parks less than two hectares (see also Fernández-Juricic, 2000). The number of plots per park was determined by park size (log-transformed). We recorded the following: cement cover; grass cover; bare-ground cover; bush cover; total tree cover; coniferous tree cover; deciduous tree cover; mean tree height; mean bush height; number of tree species; number of bush species; and number of tree stems in four diameter-at-breast-height (dbh) ranges (< 10 centimeters dbh, 10–30 centimeters dbh, 30–50 centimeters dbh, and > 50 centimeters dbh). Cover variables were visually estimated in percentages following Prodon and Lebreton (1981) and corresponded to different vegetation substrates. Tree-cover measurements were based on the area of the overstory. The number of stems was determined by counting stems in each size category in the sampling plots. Tree and bush heights were estimated with a pencil by visually rotating the plant tips 90 degrees onto the ground and then measuring the ground distance with a meter tape (± 0.05 meters). Values for each vegetation trait measured at each transect were averaged for each whole park.

We also recorded the number of pedestrians (walking and sitting) in the morning (7:00 a.m. to 12:00 p.m.) and in the afternoon (12:01 p.m. to 7:00 p.m.) in five-minute periods within 50-by-50-meter plots placed randomly inside the bird transects (one plot per transect). Measurements were conducted twice at each park (once during a weekday and once on the weekend). Final figures were averaged over the two visits and transformed into mean numbers of pedestrians/5 minutes/10 hectares per park (see also Fernández-Juricic, 2004).

**Spacing Behavior**

During the surveys, we also mapped the location of male house finches singing in each park inside and in between transects to estimate nearest neighbor distances (the distance between a male and its closest neighbor), following Krebs (1998) and Forsman, Mönkkönen, Inkeröinen & Reunanen (1998). Mapping was done using a handheld GPS device and visual landmarks. Distances were calculated using ArcView GIS software (version 3.3) and corroborated with a web-based distance calculator available online at www.wcrl.ars.usda.gov/cec/java/lat-long.htm. We estimated neighbor distances within a particular visit and then calculated the mean nearest neighbor distances over all visits per park. We included in the analysis the mean neighbor distances of house finch males in only 14 of the 35 urban parks (one mean value per park), because (a) house finches were not detected at all the parks, and (b) we only calculated neighbor distances when at least two house finches were present in the same sampling day in a park (thus avoiding temporal biases in the GPS position estimates). We also counted the number of neighbor males around the focal male in a 50-meter radius, as the density of individuals could affect patterns of spacing behavior (Krebs, 1998). As explanatory variables, we took into account the habitat structure factors that significantly explained the probabilities of park occupation: total tree cover, mean tree height, and number of stems < 10 centimeters dbh, 10–30 centimeters dbh, and 30–50 centimeters dbh.

**Male Singing Rates**

We restricted our study to males singing from perches and did not consider those cases in which males sang flying (Hill, 1993). We recorded singing rates from early March to early May 2005, between 5:45 a.m. and 9:00 a.m. on days without rain or wind. Upon finding a singing male within a transect, we recorded his songs with a Sony TCM-200DV portable tape recorder and calculated the number of songs emitted per minute. We only considered recording samples with durations of at least 90 seconds and up to a maximum of 20 minutes; recording was stopped when birds left their perching trees. We included in the analyses the singing rates of 68 males (one value per male) from 17 urban parks. Singing rates were recorded only once for a given transect to minimize the probability of resampling males. For each male, we recorded the number of neighbor males singing (which was equivalent to the number of singing males in the transect), the type of perch (tree, bush, fence, or power line, etc.), portion of the tree or bush (inner, outer) on which the animal was perching (if it was perching in a tree or bush), time of year, time of day, and temperature. Since time of day and temperature were highly correlated (Pearson correlation, $r = 0.57$, $P < 0.001$), we elected to include only the latter in the analysis. We also recorded the height of the perching bird and that of the vegetation substrate, as described before, and divided them to estimate the perch-height ratio. Ratio values close to 1 indicated that a male was perched relatively high and exposed in the substrate.

**Song Structure**

Song recordings to assess house finch vocal structure were recorded in the same parks, but on different days to those on which singing rates were recorded. We used an Audio-Technica AT815b line/gradient condenser microphone to record onto a Sony portable minidisc recorder (M2-N10). Recordings were taken from early March to early May, in the 2004 and 2005 breeding seasons, between 5:45 a.m. and 9:00 a.m. on days without rain or wind. The recording level of the minidisc recorder was the same at all sites. We could not mark the males, but to minimize the chances of recording the same individual more than once, we only visited each park once, and within each park we only recorded individuals that were separated by at least 100 meters. To minimize changes in song amplitude with distance, house finch males were recorded from a distance of 6 to 7 meters (microphone to perch). The microphone was held steadily in the most direct line toward the singing male. To reduce attenuation of songs by physical barriers, only males within unobstructed view were recorded. However, there are two sources of bias in our recordings: (a) the 1-meter difference in recording distance could generate variations in amplitude up to 1.3 dB, and (b) the orientation of the bird in relation to the microphone was not recorded, though this could also affect song amplitude.

We recorded the bird songs until 15 minutes had lapsed, the male stopped singing for more than 5 minutes, or until it flew away. We recorded 5 to 20 songs per male, with a total of 44 males in 16 urban parks. We also recorded time of the year, time of the day, and temperature as potential confounding factors. Time of the day and temperature were less strongly correlated than in the singing-rate samples (Pearson correlation, $r = 0.19$, $P < 0.05$), so we decided to include both in the analysis. We also established whether or not other males were singing while we recorded focal males, as birds can vary vocal structure in response to the presence of conspecifics (Brumm & Todt, 2002; Cynx & Gell, 2004).

Male songs were digitized at 22 kHz and 16 bits and analyzed using Raven 1.2 software (Charif, Clark & Fisrup, 2004). Over each song, we measured the following in a 22 kHz range: low frequency (Hz), frequency range (Hz), number of notes, and duration (ms). Number of notes was positively correlated with song duration (Pearson correlation, $r = 0.95$, $P < 0.001$), so we presented the results of the former. We averaged all these vocal parameters for each male across his songs, so that each data point in the analysis corresponded to a different male.

While recording a male, we also recorded 5 to 10 minutes of ambient noise before, in-between, and after song bouts. Within each male's recording session, we digitized ten randomly selected 30-second segments in which the male was not singing at 22 kHz and 16 bits, recorded ambient RMS amplitude with Raven 1.2, and calculated mean values for each session in a 22 kHz range. RMS amplitude is the sum of the squared values of amplitude for a sound (Charif et al., 2004). RMS amplitude takes into account minimum and maximum amplitude values (Bradbury & Vehrencamp, 1998) and was measured with Raven 1.2 in micro Pascals (μPa) (H. Mills, personal communication, October, 2005). A similar procedure has been recently used to record and estimate ambient noise (Leonard & Horn, 2005).

### Statistical Analysis

Throughout the statistical analyses, logarithmic transformation was performed on certain variables (park area, pedestrian rate, nearest neighbor distance, number of stems < 10 centimeters dbh, number of stems of 30–50 centimeters dbh, number of songs per minute), and arcsin transformation was performed on another (perch-height ratio) to meet normality and homogeneity of variance assumptions.

To reduce the number of vegetation variables on the habitat structure data per park, we performed a principal component analysis (PCA) on the correlation matrix. Only those PCA factors with eigenvalues > 1 were selected (Kaiser criterion), and factor loadings were rotated with a varimax raw transformation.

To analyze the effects of park area, pedestrian rate, and habitat structure (PC1–PC6, see Results) on the probabilities of park occupation by house finches, we used a logistic regression with a binomial dependent variable (presence/absence) and a logit link function. Recent studies point out the relevance of using alternative model-selection criteria, such as information theoretic approaches, to model species distributions (Rushton, Ormerod & Kerby, 2004), particularly in multicausal scenarios (Stephens, Buskirk, Hayward & Del Rio, 2005). We then calculated the Akaike information criterion (AIC) of all combinations (256) of the eight independent variables studied, and chose the model with the lowest AIC value, following Burnham and Anderson (2002).

We used stepwise multiple regressions to assess the effects of total number of individuals, total tree cover, mean tree height, and number of stems (< 10 centimeters dbh, 10–30 centimeters dbh, 30–50 centimeters dbh) on nearest neighbor distances. Both backward (F to enter = 11, F to remove = 10) and forward (F to enter = 2.5) selection procedures were performed to identify the most significant factors.

We modeled the variability in singing rates with general linear models (GLM), using two categorical and three continuous independent variables: number of singing neighbor males (1, 2, and > 3), type of perch (tree/bush, and artificial—fence/power line/building), perch-height ratio, time of year, and temperature. With a subset of these data, we conducted another GLM to assess the effects of portion of the tree in which the male was perching (inner, outer), controlling for the effects of the significant factor found in the previous analysis.

We used one-way ANOVA to determine whether the presence or absence of conspecifics singing could affect the frequency and temporal parameters of focal male songs. We also performed a GLM to assess the effects of ambient noise on low frequency, frequency range, and number of notes, while controlling for the potential confounding effects of time of year, time of day, and temperature.

Statistical analyses were conducted with SPSS 13.0 and Statistica 7.0 software.

## Results

### Habitat Structure Factors

The PCA identified six factors (PC1–PC6) with eigenvalues > 1, explaining 83.4% of the variability in park habitat structure (Table 2). PC1 was associated with ground cover—with positive values indicating relatively more grass, and negative values indicating relatively more bare ground cover. PC2 was associated with tree structure—with positive values characterizing parks with greater total tree cover, higher tree height, and more stems of 30–50 centimeters dbh. PC3 was positively associated with bush height, number of bush species, and number of stems > 50 centimeters dbh. PC4 was a tree composition axis—with positive values indicating greater deciduous tree cover and negative values indicating greater coniferous tree cover. PC5 was negatively associated with cement cover, and PC6 was positively associated with number of stems < 10 centimeters dbh and number of stems of 10–30 centimeters dbh.

**Presence of House Finch Males in Urban Parks**

Park size ranged from 0.09 to 122.94 hectares (mean ± SD, 11.80 ± 25.29). Only two of the independent factors were correlated: park area and PC4 (Pearson correlation, r = – 0.35, *P* = 0.041). This means that more coniferous cover and less deciduous cover was associated with larger parks. All other correlations were nonsignificant (P > 0.236).

House finch males were found in 26 of the 35 parks studied. The logistic regression model accounting for the probabilities of park occupation with the lowest AIC (41.27) included two of the eight factors considered: park size and PC2 (B coefficients, intercept = – 1.47, park size = 1.81, PC2 = 0.90; $X^2$ = 9.21, d.f. = 2, *P* = 0.010). Thus, the probability of house finch males being present in urban parks increased with the size of the park and with tree structure (total tree cover, tree height, and the number of stems 30–50 centimeters dbh).

**Distance Between Singing Males**

The mean (± SD) distance between male house finches within parks was 279.49 ± 374.53 meters. Neighbor distance within parks was affected by the number of stems 30–50 centimeters dbh (coefficients, intercept = – 0.073, stems 30–50 cm = 0.300; $F_{1,12}$ = 13.57, *P* = 0.003, Adjusted $R^2$ = 0.49; see Figure 1): Distance to the closest singing male increased with the number of stems of medium to large trees. This result was found in multiple regressions with both forward and backward selection procedures, which failed to include the other five factors studied: mean number of singing males in the park, tree height, total tree cover, and stems < 10 and of 10–30 centimeters dbh. A similar result was also found even after arbitrarily entering the number of singing males in the previous model (intercept, coefficient = 0.001, $t_{11}$ = 0.01, *P* = 0.992; number of males, coefficient = –0.016, $t_{11}$ = –0.79, *P* = 0.441; total stems 30–50 cm, coefficient = 0.296, $t_{11}$ = 3.56, *P* = 0.004; $F_{2,11}$ = 6.89, *P* = 0.011, Adjusted $R^2$ = 0.48).

**Male Singing Rates**

The mean (± SD) singing rate of house finch males across parks was 4.36 ± 3.07 songs per minute. The singing rate of male house finches was influenced by perch-height ratio: The higher the males were in the perching substrate, the more songs per minute they emitted (Table 3, Figure 2a). Number of neighboring males, type of perch, temperature, and time of year did not exert a significant influence (Table 3). A subset of these data was then used to assess the effects of singing position in the tree. We found that males increased the number of songs per minute while perching in the outer portions of the tree or bush (Figure 2b), controlling for the significant effects of perch-height ratio (Table 3).

**Structure of Male Songs in Relation to Ambient Noise**

The vocal parameters of the songs of the studied house finch males were characterized as follows (mean ± SD): low frequency, 1,720.98 ± 129.35 Hz;

high frequency, 15,421.94 ± 1,432.91 Hz; frequency range, 15,421.94 ± 1,432.91 Hz; and number of notes per song, 20.79 ± 4.27. All our ambient noise recordings yielded a low frequency equal to 0 Hz. The mean high frequency of ambient noise was 19,750.46 ± 1,169.04 Hz, and the RMS amplitude was 1,734.32 ± 1,112.59 µPa. The two most important noise sources in our study area were car traffic and air traffic.

We first assessed whether the presence or absence of conspecifics would affect frequency and temporal vocal parameters of house finch songs. We found that none of these parameters was affected by the social context (low frequency, $F_{1,42} = 3.09$, $P = 0.086$; frequency range, $F_{1,42} = 0.94$, $P = 0.338$; number of notes, $F_{1,42} = 0.06$, $P = 0.803$).

However, male house finches changed the frequency and temporal structure of their songs in relation to ambient noise. Controlling for the effects of time of the year, time of the day, and temperature, we found that house finch males increased the low frequency of their songs in areas with higher ambient noise (Table 4, Figure 3a). However, the frequency range of songs did not vary with ambient noise (Table 4). We also found changes in the temporal structure of songs: the number of notes per song decreased with increasing ambient noise (Table 4, Figure 3b).

## Discussion

Our results show that male house finches select relatively large urban parks with high availability of medium- to large-size trees, increase their neighbor distances with an increase in the number of stems, increase singing rates in the most exposed parts of the perch (upper and outer portions), and change the low frequency and number of notes of their songs in relation to high ambient noise levels.

The higher chances of occupation in larger parks could be the result of the reported association between park area and coniferous cover (PC4). Although the house finch does not appear to be particularly associated with coniferous forests in the western part of its distributional range (Hill, 1993), it uses open coniferous forests at high elevations (Grinnell & Miller, 1944) and prefers to nest in conifers in Ontario (Graham, 1988). It seems that male house finches in our study area prefer large parks with greater coniferous cover, probably because the thicker coniferous vegetation provides better nesting substrates. Furthermore, singing house finch males occupy parks with taller and denser vegetation (i.e., a high availability of tree cover, tall trees, and a large number of stems 30–50 centimeters dbh). This might indicate a preference for nesting sites that are less vulnerable to predation and human disturbance and have a higher availability of insects to feed nestlings.

We also found that the number of neighbor males did not affect neighbor distance or singing rates—a confirmation of earlier reports that the singing behavior of this species is not greatly influenced by competitor presence (Thompson, 1960). However, the higher the availability of stems of 30–50 centimeters dbh, the greater the distance between neighbor singing males. Although this species does not defend large territories and can nest in loose colonies (Hill, 1993; 2002), our result could be interpreted in terms of better nesting and foraging resources found in areas with denser vegetation, as explained above. Alternatively, in parks with fewer stems 30–50 centimeters dbh, the perceived risk of predation might be higher due to reduced availability of protective cover, and house finches might be decreasing neighbor distance in order to dilute that risk (e.g., Forsman et al., 1998).

The increase in singing rates by males may be associated with higher mating probabilities. Previous studies show that female house finches show preference for more colorful males and males that emit songs at faster rates (Hill, 1990; Nolan & Hill, 2004), and that colorful males nest earlier (Hill, Nolan & Stoehr, 1999) and have higher nesting success (McGraw, Stoehr, Nolan & Hill, 2001). Singing rates increased in the most exposed portion of perches. When perching in trees, male house finches may display at higher areas in the trees if these areas are of better quality for breeding—thereby increasing their chances for reproductive success. Another interpretation, which particularly applies to artificial perches (fences, power lines, buildings), is that males may increase their visual and acoustic exposure to females by singing from perches that are more easily detected from the distance, as found in golden-winged warblers *Vermivora chrysoptera* (Rossell, 2001). However, being more exposed could also attract more predators, or it could increase the chances of early detection of a predator through improved antipredator vigilance (Krams, 2001). Future studies should establish the trade-offs between breeding success and predation risk for house finch males singing from exposed perches.

Another shortcoming of singing fromexposed positions in urban areas is that house finch males may face greater acoustic disturbance. There are two types of variations to counteract the masking effects of higher noise levels, which are usually concentrated on low frequencies in cities: amplitude shifts (the Lombard effect; e.g., see Cynx, Lewis, Tavel & Tse, 1998; Manabe, Sadr & Dooling, 1998; Pytte, Rusch & Ficken, 2003; Kobayasi & Okanoya, 2003; Brumm, 2004), and frequency shifts (Slabbekoorn, 2004). Although we did not report results on song amplitude in relation to ambient noise, we found that house finch males modified some song-structure parameters, corroborating previous laboratory and field studies in other species (Sikiba, 2000; Brumm & Todt, 2002; Lohr, Wright & Dooling, 2003; Leonard & Horn, 2005).

House finch males probably raised the low frequency of their songs to minimize noise masking. This type of response has also been found in great tits (*Parus major*) in the city of Leiden, the Netherlands (Slabbekoorn & Peet, 2003). Narrower bandwidths would also decrease sound masking (Dubois & Martens, 1984; Rheindt, 2003), however, house finch males did not modify song-frequency range. A novel finding of our study was that the number of notes per song decreased rather than increased (see Lengagne, Aubin, Lauga & Jouventin, 1999) with ambient noise. This finding was rather surprising because (a) songs with more notes would increase signal detection, and (b) female house finches prefer males with long songs (Nolan & Hill, 2004). Reducing song length would be expected to have a negative effect on male mating success. One interpretation is that there could be a trade-off between song amplitude and number of notes per song to optimize energy expenditure during the breeding season. Energy could then be allocated to produce louder songs or longer songs, depending upon ambient noise levels. This explanation assumes that singing entails significant energy costs (Oberweger & Goller, 2001), but recent evidence shows that those costs may be minimal (Ward, Speakman & Slater, 2003). Alternatively, we may have found males of low quality with short songs in low-quality (e.g., noisy) areas. However, this explanation is limited by the fact that we were not able to capture males and control for body condition effects and that our amplitude estimates were constrained by some confounding factors (see Methods).

We conclude that house finch males select parks and perches with characteristics that will increase their mating success in urban areas, and change their singing behavior to minimize acoustic constraints due to high noise levels. These behavioral changes are examples of some of the flexible mechanisms this species uses to adapt to urban environments.

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## Glossary

**Akaike information criterion (AIC):** Statistical model fit measure that quantifies the relative goodness-of-fit of various previously derived statistical models, given a sample of data. The driving idea behind the AIC is to examine the complexity of the model together with goodness of its fit to the sample data, and to produce a measure which balances between the two. For more information, visit http://en.wikipedia.org/wiki/Akaike_information_criterion

**ANOVA (analysis of variance):** Statistical method that yields values that can be tested to determine whether a significant relation exists between variables.

**Arcsin transformation:** Statistical technique whereby each observation or raw value of data is replaced by the arcsin(SQRT) of itself.

**Carotenoid:** One of a group of yellow, orange, and red lipid-soluble pigments found in all chloroplasts, cyanobacteria, and some bacteria and fungi, and chromoplasts of higher plants (*Penguin Dictionary of Biology*).

**Conspecifics:** Individuals that are members of the same species.

**Diameter at breast height (dbh):** Tree diameter measured at 4.5 feet above the forest floor on the uphill side of the tree.

**Edge species:** Species adapted to habitat edges, such as those abutting forests. They are often generalist species that also adapt well to human-dominated environments.

**Eigenvalues:** The components derived from the data that represent the variation in the original data accounted for by each new component or axis (*Oxford Dictionary of Ecology*). For more information, visit http://en.wikipedia.org/wiki/Eigen_value.

**General linear model (GLM):** A statistical linear model that incorporates a number of different statistical analyses, such as, ANOVA, ANCOVA, MANOVA, linear regression, t-test, etc.

**Logarithmic transformation:** Statistical technique whereby each observation or raw value of data is replaced by the log (base 10) of itself.

**Logistic regression:** A statistical method used to predict a discrete outcome, such as group membership, from a set of variables that may be continuous, discrete, dichotomous, or a mix of any of these.

**Low frequency:** The lower frequency bound of a song (Charif, R.A., Clark, C.W. & Fisrup, K.M. (2004). *Raven 1.2 User's Manual*. Ithaca, NY: Cornell Laboratory of Ornithology.).

**Low abundant species:** Species with low local abundances.

**Interior species:** Species adapted to habitat interiors, such as interiors of forests.

**Principal component analysis (PCA):** A multivariate analysis technique that orders a set of objects in any number of dimensions (fewer is better). It involves Eigen analysis of a correlation matrix.

**Transect:** A line used in ecological surveys to provide a means of measuring and representing graphically the distribution of organisms (*Oxford Dictionary of Ecology*).

**Varimax raw transformation:** This is a rotational strategy in Principal Component Analysis aimed at obtaining a clear pattern of loadings, that is, factors that are somehow clearly marked by high loadings for some variables and low loadings for others (StatSoft, Inc. (2005). STATISTICA, version 7.1. www.statsoft.com).

**Figure 1. Relationship between the distance to the closest singing male (first neighbor distance) and number of stems 30–50 cm dbh within urban parks in Long Beach, California.**

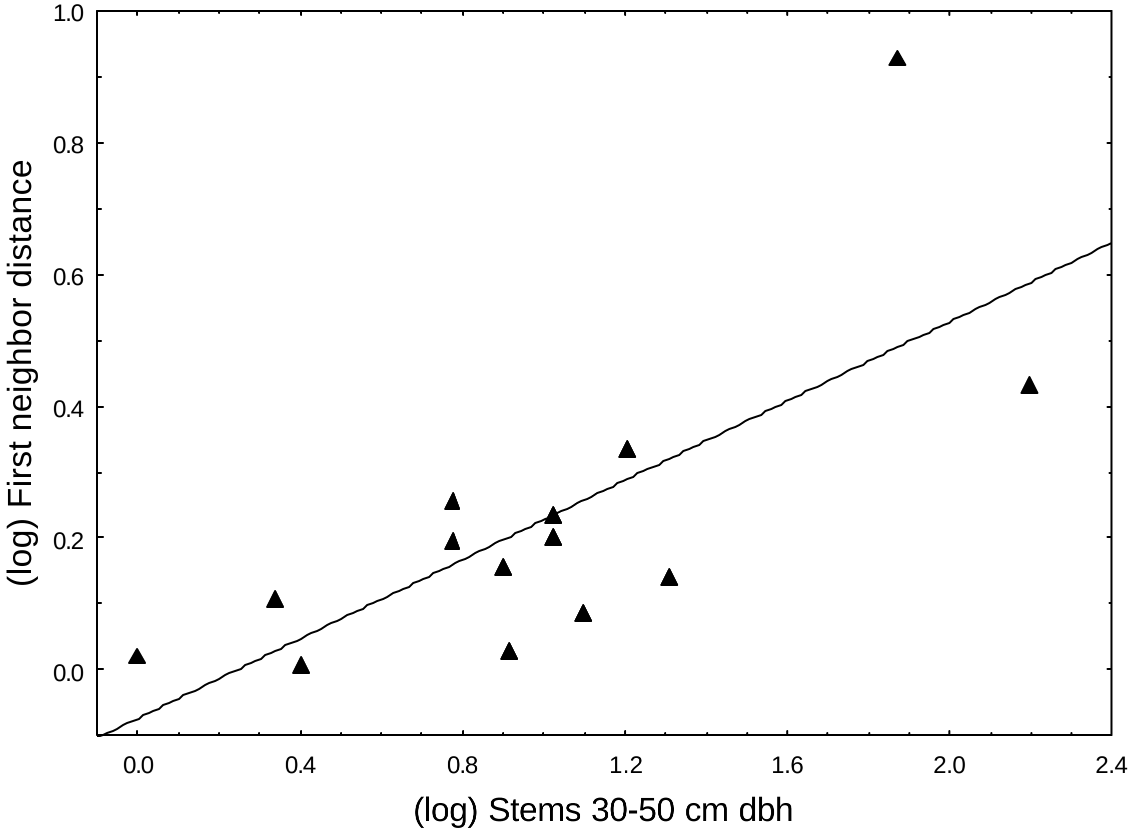

**Figure 2. Relationship between (log) house finch male singing rate and (a) (arcsin) perch height ratio (indicating how high in the perching substrate the male was singing), and (b) male position in the tree while singing (inner or outer part).**

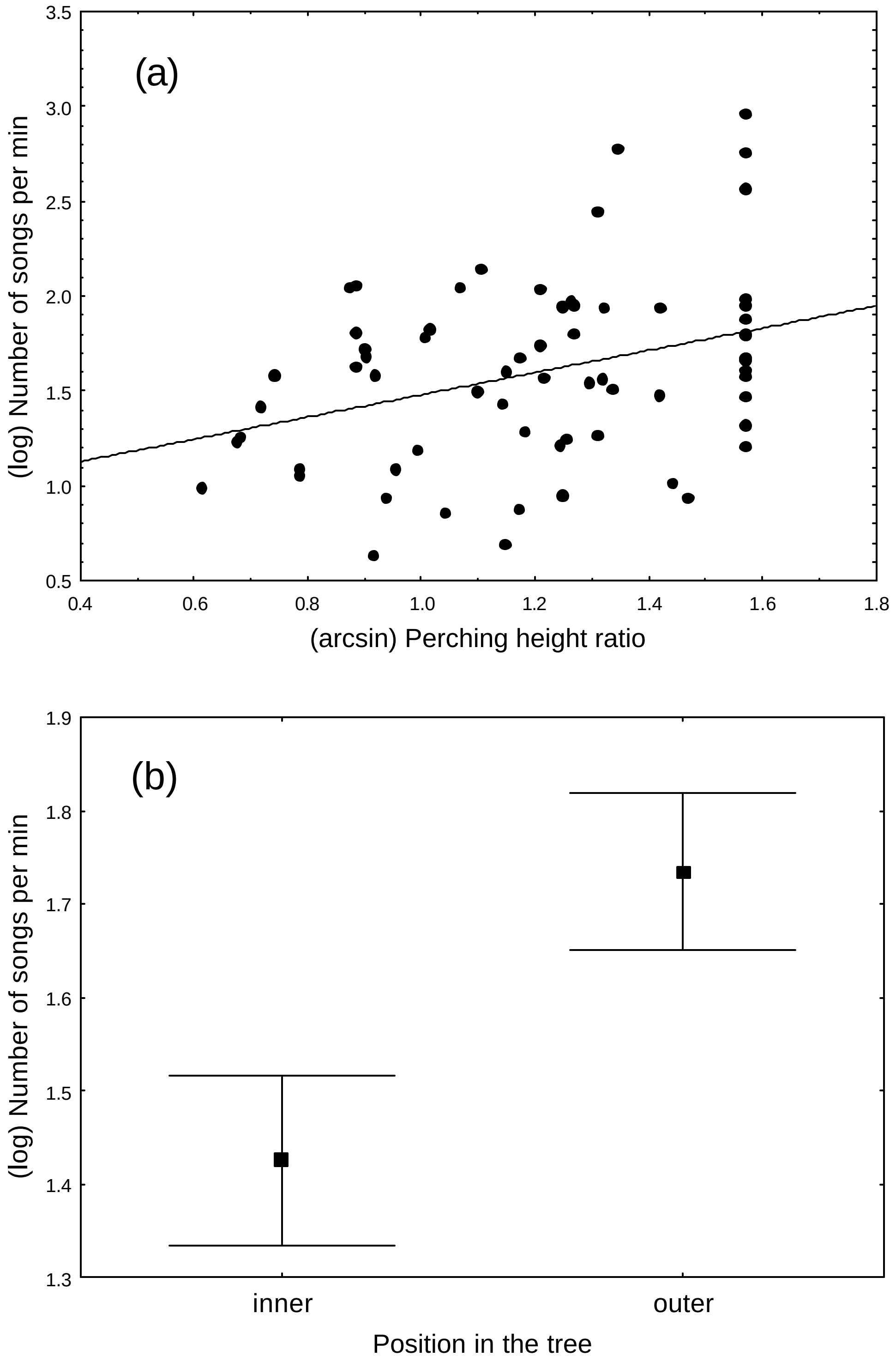

**Figure 3. Relationships between ambient noise (ambient RMS amplitude) and (a) low frequency and (b) number of notes per song.**

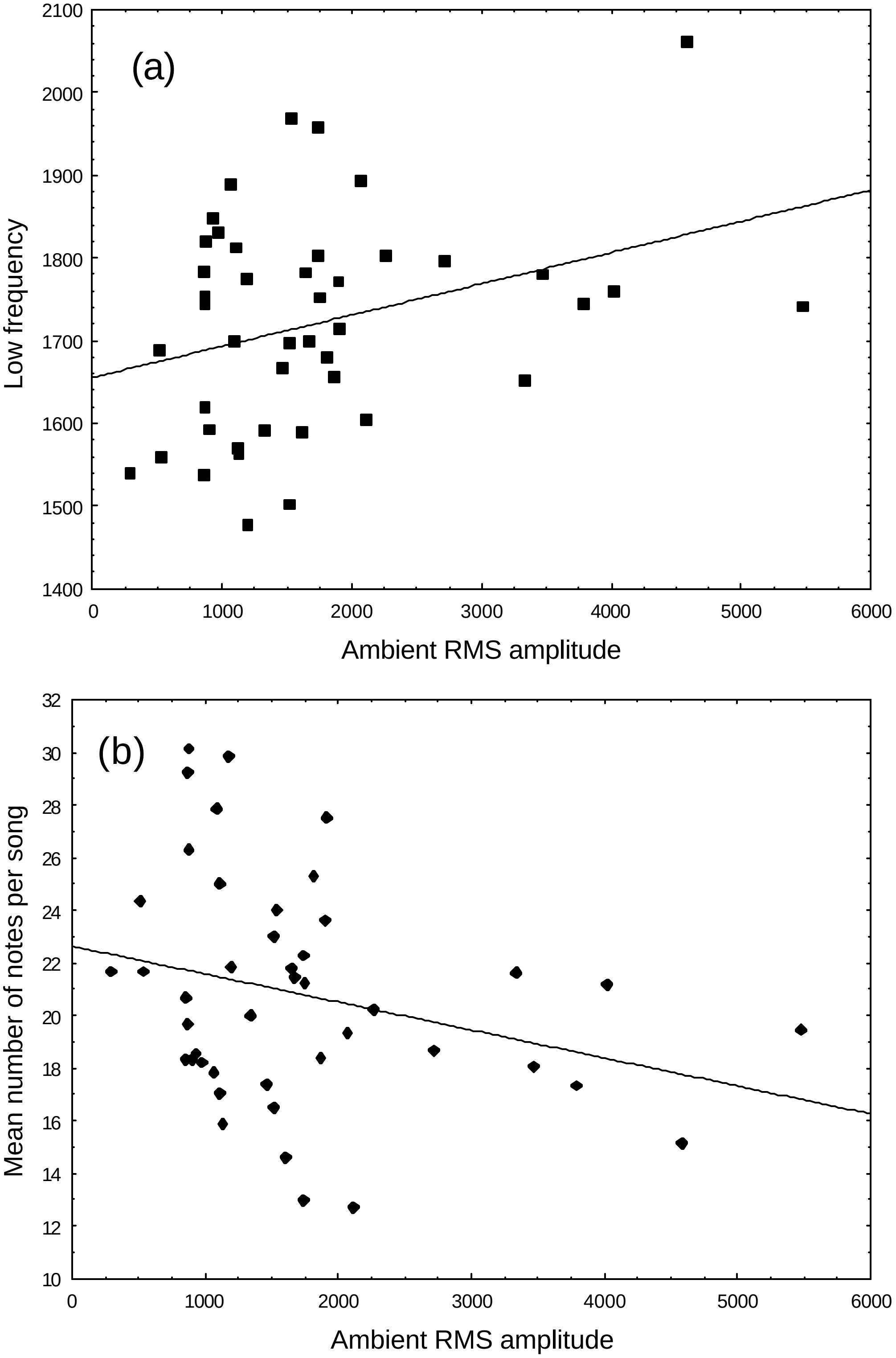

**Table 1. List of the urban parks used in this study and their sizes.**

| **Park** | **Area (ha)** |
|---|---|
| Alamitos Park | 0.09 |
| Rose Park | 0.23 |
| Bouton Creek Park | 0.32 |
| Birdcage Park | 0.39 |
| College Estates Park | 0.91 |
| Jackson Park | 0.95 |
| Will Rogers Mini Park | 0.97 |
| Douglas Park | 1.05 |
| Bixby Knolls | 1.48 |
| Somerset Park | 1.50 |
| MacArthur Park | 1.52 |
| Los Altos Park | 1.92 |
| Channel View Park | 2.06 |
| Drake Park | 2.51 |
| Ramona Park | 2.69 |
| Los Cerritos Park | 2.93 |
| Coolidge Park | 3.04 |
| Cherry Park | 3.40 |
| Admiral Kidd Park | 3.76 |
| Marina Green | 4.54 |
| Silverado Park | 4.57 |
| Bixby Park | 5.08 |
| Whaley Park | 5.22 |

| | |
|---|---|
| Hudson Park | 5.29 |
| Wardlow Park | 5.87 |
| Stearns Champions Park | 8.49 |
| Scherer Park | 9.25 |
| Cesar E. Chavez Park | 9.88 |
| Bluff Park | 10.44 |
| DeForest Park | 10.52 |
| Houghton Park | 10.64 |
| El Dorado Nature Center | 41.48 |
| Heartwell Park | 49.57 |
| Recreation Park | 85.35 |
| El Dorado Park West | 122.94 |

**Table 2. Factor loadings of the individual variables obtained by a principal component analysis (PCA) on the vegetation structure of 35 parks in the city of Long Beach, southern California. High correlations between the PCA and vegetation structure factors (factor loadings > 0.70) are marked in bold. Abbreviations: dbh, diameter at breast height.**

| | **Factor 1** | **Factor 2** | **Factor 3** | **Factor 4** | **Factor 5** | **Factor 6** |
|---|---|---|---|---|---|---|
| Cement cover | –0.0411 | –0.0848 | 0.0446 | 0.0265 | **–0.9378** | 0.0741 |
| Grass cover | **0.8748** | 0.0483 | –0.0459 | –0.0455 | 0.4021 | –0.2046 |
| Bare ground cover | **–0.9221** | –0.0248 | –0.1369 | 0.0780 | 0.1629 | 0.0567 |
| Bush cover | –0.4072 | 0.1050 | 0.6026 | –0.1404 | 0.0566 | 0.5158 |
| Total tree cover | –0.3120 | **0.7669** | 0.2959 | 0.0092 | 0.1716 | 0.0355 |
| Coniferous cover | 0.0765 | 0.1412 | 0.1119 | **–0.8501** | 0.1188 | –0.0495 |
| Deciduous cover | –0.0302 | 0.2245 | 0.2525 | **0.8416** | 0.0766 | 0.1247 |
| Tree height | 0.1975 | **0.8365** | 0.1775 | –0.0145 | –0.0012 | –0.0621 |
| Bush height | 0.0495 | 0.0459 | **0.9047** | 0.1572 | –0.1416 | 0.0533 |
| Number of tree species | 0.1496 | 0.6157 | 0.3196 | 0.3147 | 0.1542 | 0.3782 |
| Number of bush species | 0.0011 | –0.0327 | **0.7492** | 0.1572 | –0.3034 | 0.4309 |
| Number of stems < 10 cm dbh | –0.3337 | –0.0170 | 0.1250 | 0.0847 | 0.1386 | **0.8614** |
| Number of stems 10–30 cm dbh | 0.0503 | 0.1159 | 0.0922 | 0.1264 | –0.2923 | **0.8614** |
| Number of stems 30–50 cm dbh | –0.0018 | **0.7628** | –0.3449 | –0.0346 | –0.0333 | 0.0711 |
| Number of stems > 50 cm dbh | 0.1421 | 0.2072 | **0.7350** | –0.1107 | 0.3380 | –0.0056 |
| Proportion of explained variance | 0.139 | 0.160 | 0.183 | 0.110 | 0.098 | 0.144 |

**Table 3. Results from a general linear model with male house finch singing rate (log) as dependent factor, and two data subsets with different independent factors: (a) number of neighboring males, type of perch, temperature, and perch-height ratio (indicating how high in the perching substrate the male was singing), and (b) perch-height ratio and singing male position in the tree (inner or outer part).**

| | **F** | **d.f.** | **P** |
|---|---|---|---|
| **(a)** | | | |
| Intercept | 0.86 | 1, 61 | 0.357 |
| Number of neighboring males | 0.39 | 2, 61 | 0.676 |
| Type of perch | 0.19 | 1, 61 | 0.664 |
| Temperature | 0.04 | 1, 61 | 0.851 |
| Time of year | 0.87 | 1, 61 | 0.354 |
| Perch-height ratio | 8.32 | 1, 61 | **0.005** |
| **(b)** | | | |
| Intercept | 11.05 | 1, 54 | **0.002** |
| Perch-height ratio | 8.08 | 1, 54 | **0.006** |
| Position in the tree | 6.01 | 1, 54 | **0.017** |

**Table 4. Results from a general linear model assessing the effects of ambient noise RMS amplitude, and controlling for the effects of time of day and temperature on house finch low frequency, frequency range, and number of notes per song.**

| | **Coefficient** | **$F_{1,39}$** | **P** |
|---|---|---|---|
| ***Low frequency*** | | | |
| Intercept | –28368.9 | 1.14 | 0.291 |
| Noise RMS amplitude | 0.01 | 4.59 | **0.038** |
| Time of year | 0.80 | 1.25 | 0.269 |
| Time of day | 36.10 | 2.87 | 0.098 |
| Temperature | 3.90 | 0.50 | 0.481 |
| ***Frequency range*** | | | |
| Intercept | 235419.10 | 0.54 | 0.465 |
| Noise RMS amplitude | –0.20 | 0.68 | 0.412 |
| Time of year | –5.70 | 0.47 | 0.496 |
| Time of day | 75.40 | 0.08 | 0.771 |
| Temperature | –34.70 | 0.27 | 0.604 |
| ***Number of notes*** | | | |
| Intercept | | 0.46 | 0.497 |
| Noise RMS amplitude | | 4.16 | **0.048** |
| Time of year | | 0.38 | 0.540 |
| Time of day | | 0.82 | 0.371 |
| Temperature | | 0.35 | 0.555 |

# Using Citizen Science in Urban Bird Studies*

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## Abstract

Long-term monitoring projects and studies designed to survey large, variable areas often face a similar challenge: data collection. Researchers can sometimes overcome this obstacle by designing studies that utilize the skills of volunteers, or citizen scientists. Citizen scientists currently play active roles in a wide range of ecological projects, and their contributions have enabled scientists to collect large amounts of data at minimal cost. Because bird-watching is popular among members of the general public, bird-monitoring projects have been among the most successful at integrating citizen scientists. Several large-scale studies, such as the Christmas Bird Count and Breeding Bird Survey, have successfully relied on citizen scientists to collect data. As urban areas expand and scientists work to find ways to manage wildlife in cities, information about the associations among animals and urban environments is needed. By utilizing the large pool of potential participants in urban areas, citizen science–based studies can play an important role in collecting this information. One such study, the Tucson Bird Count (TBC), has successfully utilized citizen scientists to collect information on the distribution and abundance of birds across an urban area. The results from the TBC have been used in numerous scientific studies, and they are helping wildlife managers identify important sites for birds within the city, as well as land-use practices that sustain native birds.

**Key Words:** birds, bird survey, citizen science, monitoring, Tucson Bird Count

## Introduction

The value of employing volunteers from the general public (i.e., citizen scientists) to collect data has been recognized for a long time, but there has been a recent surge in studies based on citizen science. Although many early citizen science programs were primarily conceived as educational tools—as a way to increase participants' knowledge about science (Brossard, Lewenstein & Bonney, 2005)—there has been a growing focus on the use of citizen scientists to collect long-term data (Brewer, 2002; Evans et al., 2005). For research projects that require many observers, such as studies designed to assess the status of local resources, establish baseline ecological measures, or identify the impacts of various activities on environmental quality, citizen scientists can be a

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remarkable resource (Ely, 2000; Altizer, Hochachka & Dhondt, 2004). In many cases, lone professional scientists, lacking the necessary funding and manpower, are unable to gather the broad-scaled yet detailed information that a cadre of citizen scientists can. As a result, citizen scientists are currently active participants in a wide range of ecological projects, including studies of macroinvertebrates and stream health (Fore, Paulsen & O'Laughlin, 2001), the status of reef fish and amphibian populations (Pattengill-Semmens & Semmens, 2003; Ebersole, 2003), and the distribution and abundance of monarch butterflies in North America (Monarch Larva Monitoring Project, 2001).

One area in which citizen scientists are widely utilized is in surveying and monitoring bird populations (Lepczyk, 2005). Several large-scale bird-monitoring projects, such as the Breeding Bird Survey (BBS), Christmas Bird Count (CBC), and Project FeederWatch (PFW) rely on volunteers to collect data. The involvement of citizen scientists in these projects provides the sponsoring agencies (BBS = the United States Geologic Survey and the Canadian Wildlife Service, CBC = the National Audubon Society, PFW = the Cornell Lab of Ornithology and Bird Studies Canada) with data that allow them to generate detailed distribution maps for species throughout the United States and monitor changes in bird populations over time (Root, 1988; Sauer, 2003). Additionally, data collected through these programs have been utilized in numerous scientific studies (see LePage & Francis, 2002; Sauer, Hines & Fallon, 2003; Dunn et al., 2005).

The benefits of incorporating citizen science into ecological research are not limited to the quantity of data that can be collected. Citizen science projects can also benefit the volunteers and the broader community. Volunteers not only gain experience in making observations and participating in a scientific study, they may also develop a greater sense of stewardship over the populations or sites they are responsible for surveying or monitoring (Carr, 2004). The broader community benefits from an increased sense of stewardship among the citizen scientists and from the fact that the data collected can serve to inform local planning and land-use decisions (Nerbonne & Nelson, 2004). In addition, when the results of citizen science projects are made accessible to the public, they can help to increase residents' knowledge and appreciation of both their local ecological resources and the scientific process (Brewer, 2002).

Despite the value of volunteers to research programs and the potential benefits to both participants and the broader community, citizen science–based projects have been criticized for lacking scientific rigor (Irwin, 1995). In order to encourage the participation of volunteers, scientists design programs that attempt to both minimize the time and effort required of participants and maximize the output of usable data. As a result, the design of such programs is often a compromise. The CBC, for example, has been criticized because there is variability in volunteer effort among and within count circles, and because the count circles are not randomly located (Dunn et al., 2005). Lack of randomly located survey sites is an issue with many citizen science projects because sites tend to be clustered near population centers (where the majority of participants live) or in ecologically rich areas (which participants are more interested in surveying). Additional concerns about the quality of data collected by citizen scientists include variability in skill levels among volunteers, underreporting of negative results, and the impact of participants' biases on data quality (Irwin). In response to these concerns, many existing projects have developed innovative ways to standardize data collection (Dunn et al.) to make analysis and interpretation of results more reliable.

Given the increased interest in citizen science and the development of improved methods of data collection, researchers have begun to explore new opportunities for involving citizen scientists. One area in which there is significant potential for developing citizen science–based projects is in urban centers. While underrepresented in many traditional ecological surveys, studies based in urban areas have increased as more and more scientists are investigating the relationships between people and their environments (Marzluff, Bowman & Donnelly, 2001). Conducting comprehensive studies in urban areas frequently poses challenges to scientists because cities generally are large and consist of variable environments. Collecting representative information from such large, diverse areas requires considerable manpower, and in urban areas citizen scientists can help meet this need.

Among the more common types of ecological projects utilizing citizen scientists in urban areas are studies examining urban bird populations. Using information collected by residents, researchers in England have been able to investigate the distribution and geographical abundance of house sparrows (*Passer domesticus*) throughout London (Royal Society for the Protection of Birds, 2004). At urban and suburban sites throughout the Washington, D.C., area, citizen scientists are gathering data about the nesting success and survival rates of their backyard birds through the Smithsonian Institute's Neighborhood Nestwatch Program (Smithsonian Institute, n.d.). The Cornell Lab of Ornithology currently has five ongoing citizen science–based studies targeting urban birds (Cornell Lab of Ornithology, 2004). Another project that has utilized citizen scientists to collect data on bird abundances and distributions in an urban area is the Tucson Bird Count (Tucson Bird Count, n.d.).

## Methods

### Case Study: The Tucson Bird Count

The Tucson Bird Count (TBC) is a volunteer-based bird-monitoring project in Tucson, Arizona (Turner, 2003). The TBC was established in 2001 to acquire information about the distribution and abundances of birds throughout the Tucson area. Tucson has been the setting for many studies investigating the relationships among birds and urban environments (see Emlen, 1974; Tweit & Tweit, 1986; Mills, Dunning Jr. & Bates, 1989; Germaine, Rosenstock, Schweinsburg & Richardson, 1998), but each of these studies was restricted in coverage, in space, or in time. Furthermore, nationwide bird-monitoring programs, like the CBC and the BBS, do not provide detailed information on birds in Tucson. The CBC includes a count circle in Tucson, but due to the nature of this project, surveys occur in the winter, when many species that breed in or migrate through Tucson are not present (National Audubon Society, 2005). And though the BBS surveys birds during the spring breeding season, urban areas are generally avoided in count circle placement (O'Connor et al., 2000), and there are no BBS survey sites in Tucson (Sauer et al., 2005).

The long-term goal of the TBC is to help identify ways to restore and sustain native bird species (Turner, 2003). The initial design of the TBC focused on counting birds at random sites covering a wide range of environments throughout Tucson during the breeding season (the Route Program). The original sampling scheme allowed for inferences about bird/habitat associations to be drawn from the TBC data (Ramsey & Schafer, 2002), but several of the most bird-rich sites in Tucson were not included. As a result, the Park Monitoring Program was established as an additional component of the TBC. In the Park Monitoring Program, volunteers survey birds in their local parks, washes, or other areas of interest on a quarterly basis. Experience gained from designing the TBC program, recruiting volunteers, making the data available to the public and researchers, and confronting the challenges associated with such a study, can serve as an example for other researchers interested in establishing citizen science–based projects in urban areas.

### Study Design: The Route Program

The design of the TBC Route Program is modeled after that of the BBS: One survey site is randomly located within each one-square-kilometer (1 km²) cell of a grid covering the Tucson area, following a stratified random sampling design (for a detailed description, see Turner, 2003). Adjacent sites are grouped into routes, with an average of ten sites per route. Using maps and information available at the TBC website, volunteers select a route (or routes) to survey. (Once a route is adopted, it is unavailable to other volunteers.) Each year between April 15 and May 15 (the peak breeding season in Tucson), volunteers survey the sites along their route on a morning of their choice, conducting a five-minute unlimited-radius point count (per Blondel, Ferry & Frochet, 1981) at each site.

In 2001, 661 randomly located sites on 63 routes in the Tucson area were surveyed (Tucson Bird Count, n.d.). With the exception of sites that have been relocated due to changes in accessibility or other disruptive factors, site locations are permanent, allowing for the same sites to be surveyed each year. Tucson has grown by more than 30 square miles since 2000 (City of Tucson, 2005), and additional sites have been added to the Route Program in newly developing areas around the edges of the city. In 2005, 772 sites on 72 routes were surveyed (Tucson Bird Count, n.d.; yellow dots in Figure 1). Due to the volunteer-based nature of the TBC and the fact that participants select the routes they want to survey, there is no guarantee that all routes will be adopted every year. As a result, each year a few routes are not surveyed. To avoid gaps in coverage, routes that were not surveyed one year are given priority the following year, or are surveyed by TBC staff.

### Study Design: The Park Monitoring Program

Locations monitored in the Park Monitoring Program (red stars in Figure 1) were selected by TBC personnel and volunteers and include neighborhood, city, and regional parks, washes, a national park, and other key birding spots (referred to as parks here). The number of survey sites per park varies from 3 to 14, depending on the size of the site and the variability of environments contained within the park. To gain a better understanding of how Tucson's bird populations change throughout the year and to detect species that may not be present during the Route Program survey period, parks are monitored four times a year: winter (January 15–February 15); spring (April 15–May 15), covering spring migration/breeding; summer (July 1–31); and fall (September 1–30), covering fall migration.

Volunteers use a combination of point counts and transects to survey birds at park sites (Tucson Bird

Count, 2005). To make sites monitored in the park and Route Program comparable, five-minute unlimited-radius point counts are conducted at most sites. However, at sites with relatively open and uniform environments (Sutherland, Newton & Green, 2004), or at sites where the structure of the vegetation is more conducive to the use of line transects (e.g., a wash bordered by riparian vegetation), participants survey 200-meter transects for ten minutes (Bibby, Burgess & Hill, 1992). To reduce the chances of double-counting birds, point-count locations are separated by at least 250 meters, and transects do not cross each other (Sutherland et al.). Currently, the TBC surveys 178 sites in 21 parks in the Tucson area through its Park Monitoring Program.

**TBC Participants**

The TBC is performed primarily by volunteer birders from the Tucson community. In order to ensure that data collected in the TBC is of high quality, participants must be knowledgeable birders. To participate in the TBC, volunteers must meet the project's definition of a skilled observer, defined as "one who can identify the 25 most common Tucson-area species quickly by sight or sound, is familiar with most other birds of the Tucson area, and may need quick reference to a field guide for certain less-common or difficult-to-separate species" (Turner, 2003). To determine if participants meet this criteria, volunteers are required to take a self-test at the TBC website prior to adopting a route.

The presence of a large group of active and experienced birders in the Tucson area has been key to both the establishment and continuation of the TBC. At the initiation of the count, in 2001, volunteers were recruited through the Tucson Audubon Society newsletter, the Arizona/New Mexico birding e-mail listserv, and personal communications between TBC staff and local birders (Turner, 2003). Since 2001, new volunteers have been recruited through presentations about the TBC by project personnel, articles in the local media, and referrals from existing volunteers. In 2001, 51 primary observers and 30 additional observers participated in the Route Program, and 7 primary observers and 6 additional observers participated in the Park Monitoring Program. In 2005, 58 primary observers and 29 additional observers participated in the Route Program, and 21 primary observers and 18 additional observers participated in the Park Monitoring Program.

To decrease the time and effort required of TBC personnel in recruiting new volunteers, an emphasis is placed on retaining skilled volunteers. Since 2001, an average of 79% of volunteers have returned from year to year. This relatively high rate (Bradford & Israel, 2004) is achieved by following several known principles of volunteer retention: creating motivation, providing options, and maintaining connectedness (McCurley & Lynch, 1997). The motivation for volunteers to initially and persistently participate in the TBC is in many cases related to the goal of collecting data that will be used to help sustain native birds in Tucson. All TBC participants are skilled birders, thus a project designed to aid in bird conservation and identify the distribution and abundances of birds throughout Tucson is closely aligned with the interests of the participants.

Providing people with options that meet their level of interest in the program also aids in volunteer retention. For example, TBC participants can choose to take part in either (or both) the Route Program or the Park Monitoring Program, depending on their interests. Additional options are offered to participants as they select their route or park to survey, and they can change their selections from year to year. Participants changing the areas they survey could potentially reduce the consistency of results, but excessive changes have not happened. Rather, allowing participants this freedom and flexibility of selection has helped them find areas they are interested in monitoring long-term.

Finally, connectedness is maintained between TBC personnel and project participants in several ways. First, the results submitted by each TBC participant are available immediately and publicly on the TBC website, so volunteers are able to see how their results fit into the overall project. Second, via e-mail, articles in the Tucson Audubon Society newsletter, and a recently established annual newsletter, participants are regularly updated about the TBC's results and how they are being used.

**TBC Website**

The TBC website was designed to facilitate participants' involvement in the project, make the data collected through the project easily accessible to the public, and provide information about the project to interested parties. At the website, participants can register for the TBC, take the required self-test, view a map of available routes, select a route or park to survey, enter their data, and view results. Enabling participants to carry out these administrative tasks themselves reduces the burden on TBC personnel and speeds up the process. An additional benefit of participants entering their data via the website is a decreased risk of transcription mistakes, as participants themselves (as opposed to TBC personnel) are entering data that they recorded. The data is automatically recorded in the TBC's digital database, allowing participants to immediately review their results (and make corrections, if necessary). To further ensure the validity of the data, TBC staff review all count results submitted, correct obvious errors, and contact participants to verify any unusual or unexpected observations.

Results entered by participants are publicly available in real time on the TBC website in tabular format and as distribution maps. As a result, during each survey period, the tables and maps are constantly updated as participants enter new data. By clicking on any of the survey sites on a distribution map, users can view which other species were observed at the site. The results and distribution maps from previous Route and Park Monitoring program surveys can also be viewed at the website, allowing users to see how the distributions or abundances of various species have changed over time. An additional tool available on the TBC website is the Bird ID Center. At this page, users can view photos, hear the calls, and read an identification description from the USGS Patuxent Bird Identification Center (Gough, Sauer & Iliff, 1998) for more than 150 species found in the Tucson area.

The results and information available at the TBC website are also intended to inform researchers and local decision makers about the TBC and the data available from it. One of the advantages of having the TBC data in a digital database is that this data can be easily accessed by other users (Dunn et al., 2005). The TBC website contains a listing of projects, presentations, and publications that have made use of the TBC data so far.

## Results

### General Species Distribution Patterns

Data collected by citizen scientists through the TBC has enabled the generation of detailed Tucson-area distribution maps for more than 200 bird species. These maps indicate that many species follow distinct distribution patterns in Tucson. A group of native desert birds, exemplified by Gambel's quail (*Callipepla gambelii*), have similar distributions, with relatively high numbers of individuals on the less developed edges of Tucson, but few individuals occupying the city's more urbanized center (Turner, 2003; Figure 2). Rock pigeons (*Columba livia*), on the other hand, are among a group of species that exhibit the inverse pattern, reaching their highest abundances toward Tucson's urban core (Figure 3). Other distribution patterns are apparent in the data, including the restriction of some species to riparian areas and the concentration of others in areas of relatively dense native woodlands (Turner, 2003). Some bird species, such as mourning doves (*Zenaida macroura*), Gila woodpeckers (*Melanerpes uropygialis*), and house finches (*Carpodacus mexicanus*) have been counted in relatively high numbers at almost every survey point (Figure 4). Yet species more sensitive to development, such as the rufous-winged sparrow (*Aimophila carpalis*), are found in relatively small numbers and only at the very edges of the city (Figure 5).

### Differences Between the Route and Park Monitoring Programs

Since the initiation of the TBC in 2001, participants have recorded more than 164,000 birds representing 212 species. A total of 158 species have been recorded through the Route Program, and a total of 202 species have been recorded through the Park Monitoring Program. Fifty-four species observed through the Park Monitoring Program have not been counted through the Route Program. This difference is largely attributable to two factors. First, Tucson has a semiarid climate, and wetlands are relatively rare among the randomly located sites in the Route Program; however, several of the locations monitored in the Park Monitoring Program contain ponds, lakes, or rivers. As a result, many water-associated species, such as the American wigeon (*Anas americana*), have only been observed at park locations (Figure 6). Second, the more intensive nature of the Park Monitoring Program, with its quarterly monitoring and multiple sites per bird-rich location, has resulted in sightings of several less common species, such as the golden eagle (*Aquila chrysaetos*), Lewis's woodpecker (*Melanerpes lewis*), and Hammond's flycatcher (*Empidonax hammondii*), not found in the Route Program.

Despite these differences, both programs are complimentary projects designed to provide us with a better understanding of the distribution and abundance of birds in Tucson, as well as which sites are particularly important for birds. Through the Route Program we collect data annually from hundreds of sites across the Tucson area, and this allows us to understand better the distribution of birds citywide and monitor year-to-year changes in Tucson's bird population. Through the Park Monitoring Program, we collect year-round data at several bird-rich sites, and this helps us develop a more complete picture of all the bird species that occur in Tucson and evaluate the importance of these park locations to birds.

### How the TBC Results Have Been Used

Scientists have used TBC results to assess how species respond to different types of land use (Turner, 2003), investigate the relationship between the distribution of birds and people in Tucson (Turner, Nakamura & Dinetti, 2004), and examine the effects of nonnative cavity-nesting birds on the health of saguaro cacti (*Carnegiea gigantea*; Hutton, 2005). In addition, data from the TBC have served as baseline information for a number of scientific studies, ranging from an investigation of the impact of West Nile virus on Tucson's birds to a comparison of riparian birds in Tucson and Phoenix riparian areas. The TBC data has also been used in local land-use planning (Pima County, 2004), and to evaluate potential sites for natural resource parks (Rosen & Mauz, 2001).

### Challenges and Lessons Learned

While the TBC is an example of how citizen science and urban ecological monitoring can be integrated, executing such a program poses challenges. Both components of the TBC—the citizen scientists and the urban setting—have presented obstacles over the program's five-year history. All TBC volunteers are required to take a self-test prior to registration to ensure that they meet minimum requirements, but variation remains among participants' abilities to detect birds by sound and sight. With the TBC, this hurdle is overcome in survey design (routes are arranged so that multiple observers cover any one part of Tucson) and during data analysis (patterns of distribution and abundance are analyzed at broad scales, rather than at specific points, reducing potential observer biases). The volunteer nature of a citizen science–based project also poses challenges. With the TBC, this has manifested itself in the difficulty in getting certain routes monitored—those that are deemed less "birdy" or are otherwise unattractive to participants. In Tucson, many of these routes are clustered in one region of the city, and the less frequent monitoring of these routes has led to gaps in the TBC's citywide coverage. Initial attempts to combat this problem by encouraging existing participants to adopt these routes were relatively unsuccessful. To solve the problem, future efforts to increase monitoring in gap areas will center on 1) working more directly with local residents, many of whom are minorities, to involve them in the TBC; and 2) establishing a core set of participants willing to cover the highest-priority unadopted routes each year.

All long-term monitoring projects encounter changes in the environment surrounding their monitoring sites, but projects conducted in urban settings may encounter such changes at a more rapid rate. For the TBC, such changes have often resulted in difficulty accessing established sites due to land-use changes. The conversion of public land to private land, the gating of large, residential neighborhoods, expansion of roads, and new construction have all impacted TBC sites, requiring some to be relocated. To keep site relocations to a minimum and to provide stable monitoring sites for the future, all TBC sites were recently evaluated, and sites in danger of future relocation were moved to nearby positions expected to be more permanent. Key to the changing nature of the TBC's urban setting is Tucson's ongoing expansion. In order to continue to provide citywide information on Tucson's birds, the TBC has had to expand as well, adding new routes as needed. While such expansion ensures that the TBC keeps pace with Tucson's boundaries, it also requires additional volunteers and TBC personnel time to establish the routes.

## Conclusions

The TBC demonstrates the value of utilizing citizen scientists to collect citywide data for monitoring urban birds. Currently, the TBC is the only recurring volunteer-based, citywide urban bird-monitoring project in the world (Turner, 2003). Lessons learned through the TBC about study design, volunteer management, and the importance of making results available and useful may be helpful in the establishment of other urban bird-monitoring projects. Urban areas offer enormous potential for citizen science projects, not least because a large number of prospective volunteers are already in place. The ecology of urban areas is a growing field that requires further investigation; the type and scope of information citizen scientists can provide is invaluable. In the case of bird-related projects, there is the additional advantage that many experienced birders are interested in participating in projects designed to benefit birds, and they bring an established skill set to the project.

The breadth of data that can be collected by an organized group of citizen scientists allows researchers to conduct studies that might otherwise be impossible: Patterns of distribution and abundance can be mapped on a large scale and surveys can be regularly repeated, enabling researchers to monitor changes in populations over time. Results from these studies can be invaluable in identifying key areas of ecological importance within a city and tracking how changes in land use and other environmental factors influence bird communities. As urban areas expand, determining how populations of native wildlife can be sustained in cities is becoming more important. Data collected by citizen scientists can be a vital tool in helping meet this challenge.

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## Glossary

**Point count:** A record of organisms heard or seen within a given radius of a survey site during a set period of time.

**Transect:** A line used in ecological surveys to provide a means of measuring and representing graphically the distribution of organisms (*Oxford Dictionary of Ecology*).

**Figure 1. Map of the Tucson Bird Count study area showing major washes, roads, Route Program survey sites, and Park Monitoring Program locations.**

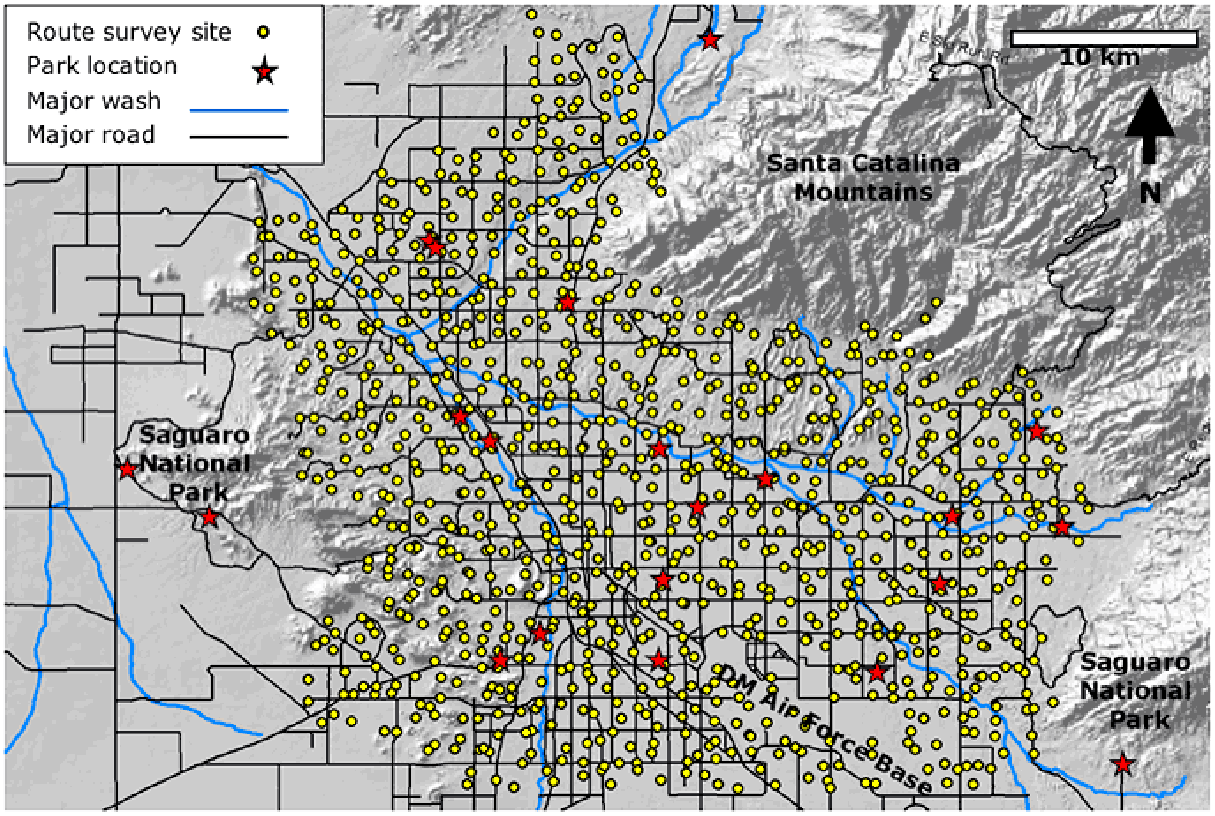


**Figure 2. Distribution and abundance of Gambel's quail across the Tucson Bird Count Route Program study area. Although actual survey site locations are randomly located within each 1-km² cell, results are shown at cell centers.**

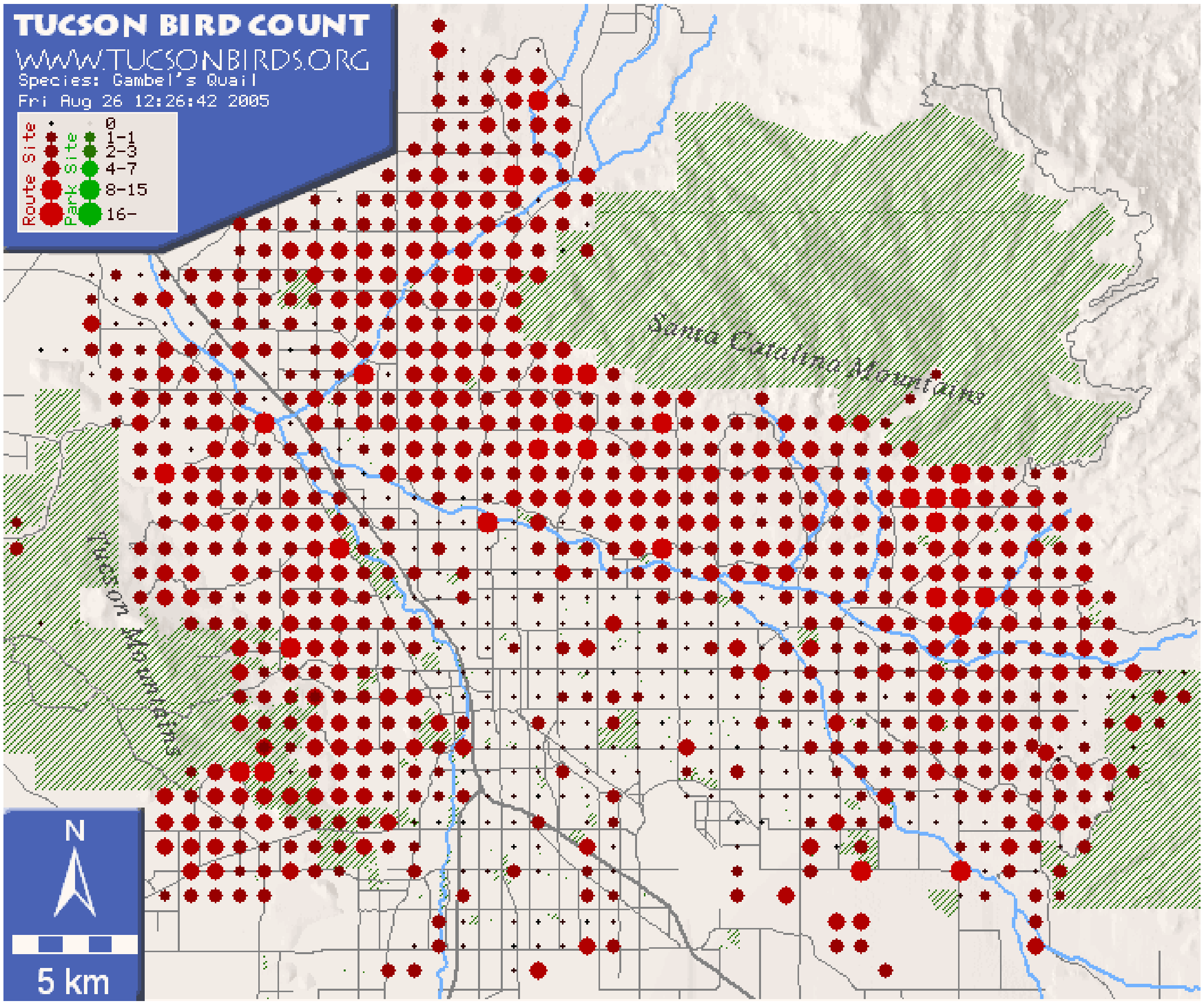


**Figure 3. Distribution and abundance of rock pigeons across the Tucson Bird Count Route Program study area. Although actual survey site locations are randomly located within each 1-km² cell, results are shown at cell centers.**

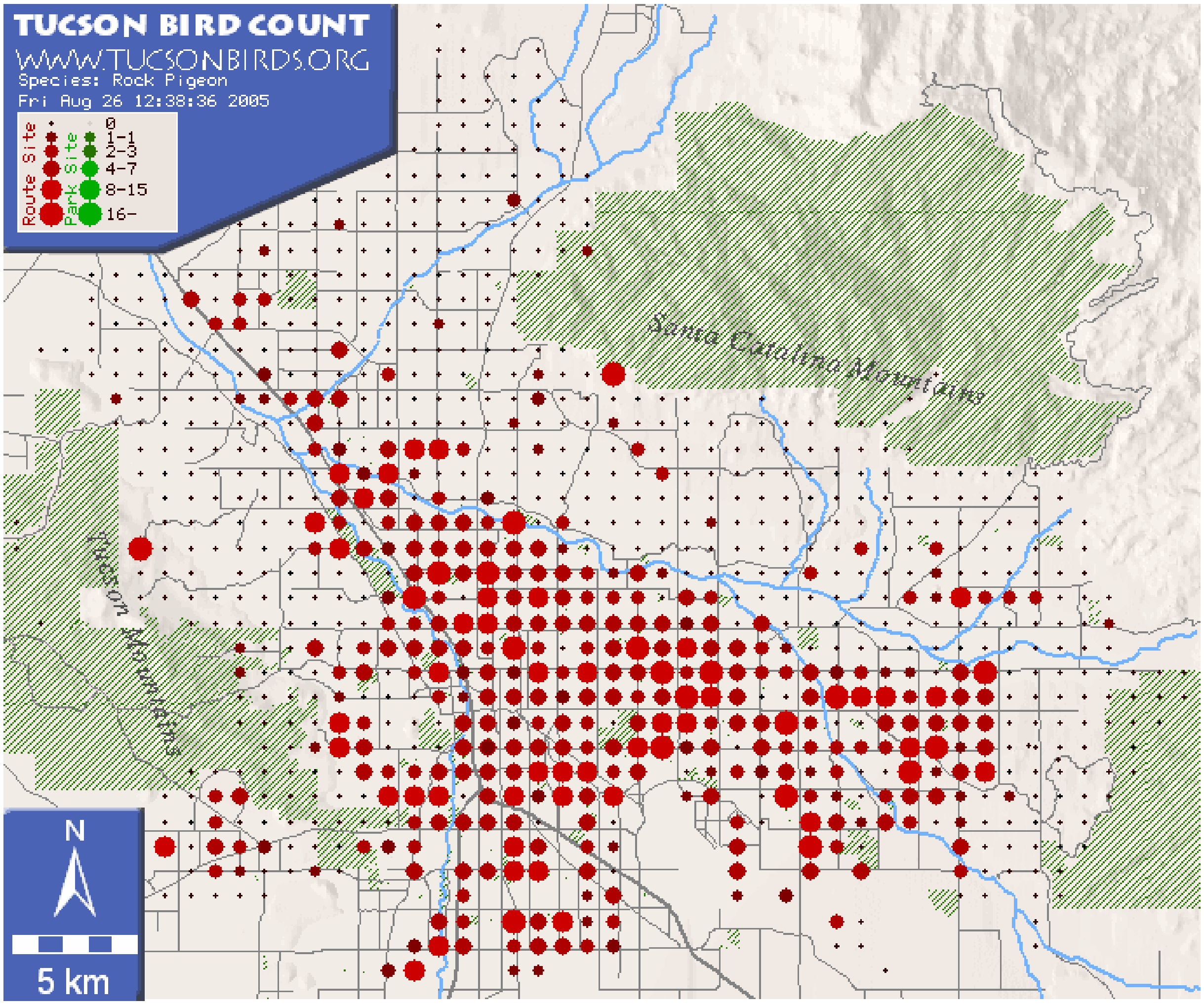


**Figure 4. Distribution and abundance of mourning doves across the Tucson Bird Count Route Program study area. Although actual survey site locations are randomly located within each 1-km² cell, results are shown at cell centers.**

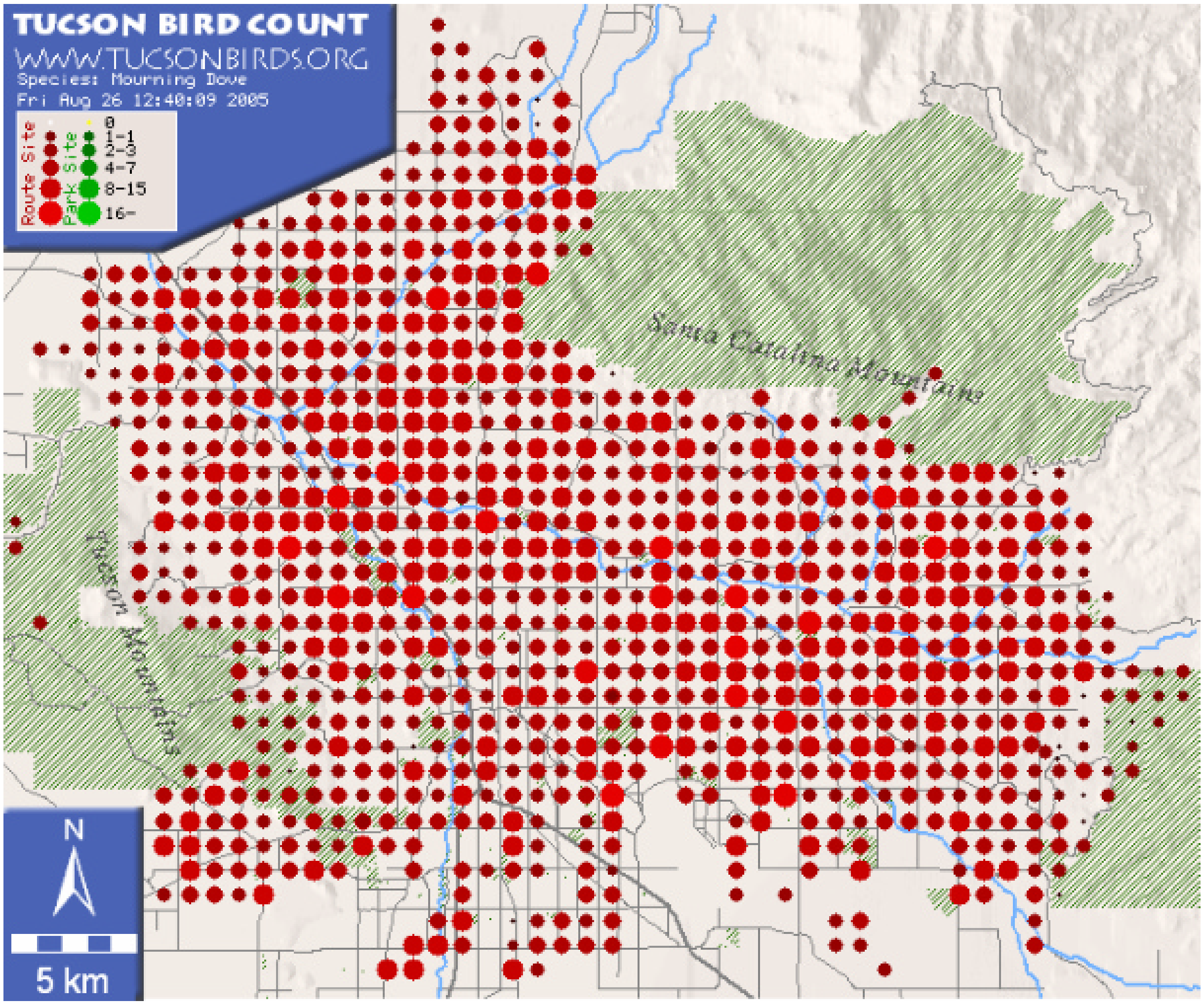


**Figure 5. Distribution and abundance of rufous-winged sparrows across the Tucson Bird Count Route Program study area. Although actual survey site locations are randomly located within each 1-km² cell, results are shown at cell centers.**

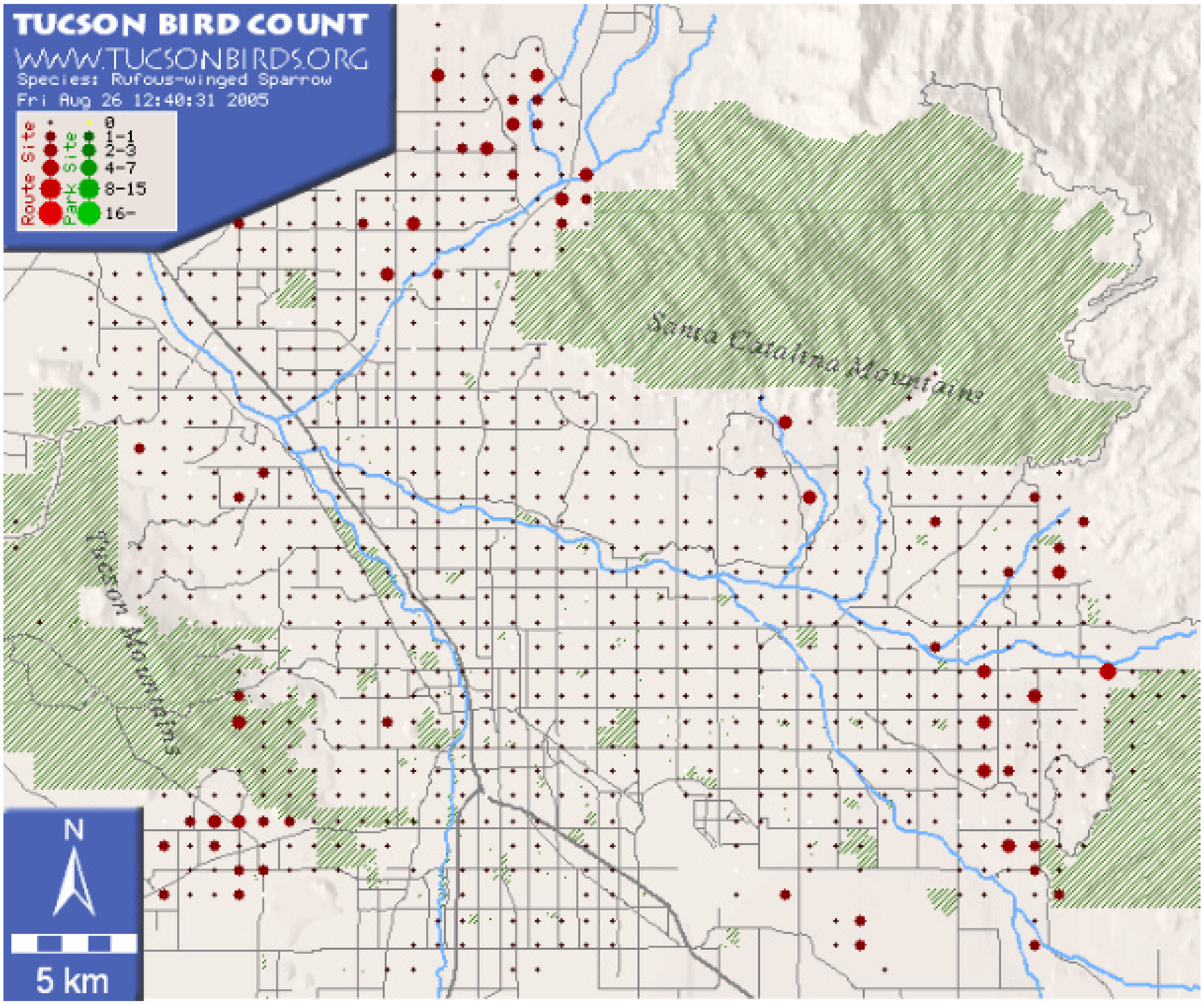
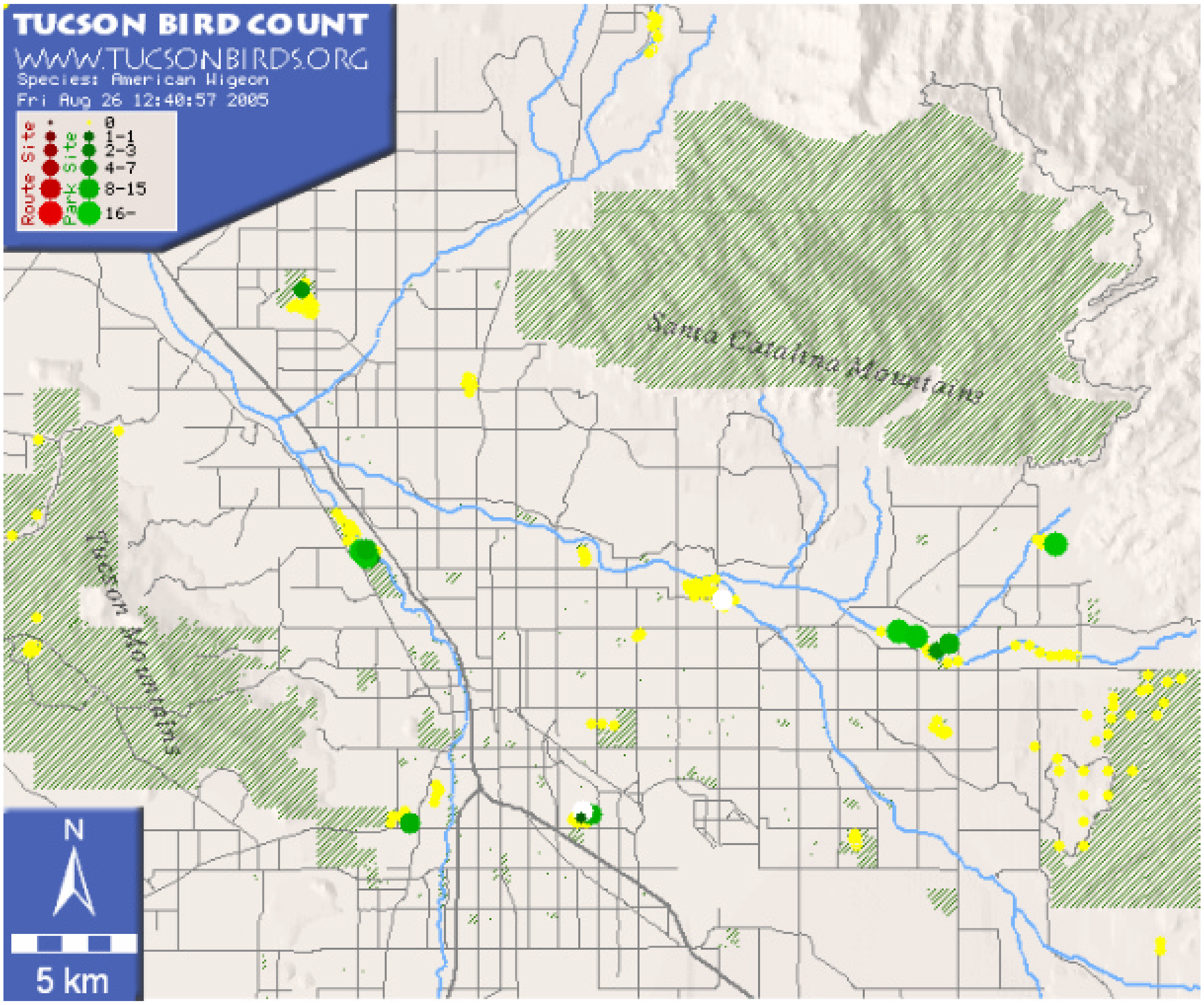


**Figure 6. Distribution and abundance of American wigeon at the Tucson Bird Count Park Monitoring Program locations. Park locations are shown in yellow, with count results shown in green.**



# Avian Response to Restoration of Urban Tidal Marshes in the Hackensack Meadowlands, New Jersey*

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## Abstract

Tidal marshes located in urbanized regions have experienced a long history of degradation. As a result, restorations have frequently been conducted to improve the habitat quality of these marshes. Few studies, however, have investigated the effect of restoration on avian community composition in urban tidal marshes. To this end, we conducted avian surveys for one year prior to restoration and three years after restoration at Harrier Meadow marsh, in the Hackensack Meadowlands, New Jersey. After restoration, avian species richness and abundance increased, while evenness decreased, mostly due to large flocks of sandpipers sporadically visiting the marsh during migration. Prior to restoration, generalists were by far the most abundant foraging guild, while they shared dominance with mudflat and open-water foragers after restoration. Avian surveys were also conducted for three years after restoration at Mill Creek marsh, also in the Meadowlands. Though the restoration goals were the same for Harrier Meadow and Mill Creek, the two marshes had distinct habitat compositions after restoration, and this allowed us to examine avian response to variation in habitat availability. In all three years of monitoring after restoration, Harrier had a greater avian density and higher species richness than Mill Creek; however, avian abundance at both marshes was dominated by the same three foraging guilds. Evenness did not differ across post-restoration years or between marshes. Avian abundance showed a decreasing trend during the three years of post-restoration monitoring; however, further monitoring will be necessary to determine the long-term trends in the avian community.

**Keywords:** avian community structure, foraging guild, Hackensack Meadowlands, marsh restoration,

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*Phragmites*, restoration monitoring, tidal marsh, urban

## Introduction

Estuarine tidal marshes serve as important foraging grounds and juvenile nurseries for a variety of fish species and crustaceans (Kneib, 1997) and provide important habitat for many migratory and resident bird species (Reinert & Mello, 1995; Burger, Niles & Clark, 1997; Melvin & Webb, 1998). Yet many coastal North American metropolitan areas, such as New York City, Boston, Seattle, and San Francisco, are located in, on, or near tidal marshes. As a consequence, urban tidal marshes have been altered for various types of development, and this has caused tidal restriction and habitat fragmentation and loss. Urban marshes have also been on the receiving end of a wide variety and high concentration of pollution from sources such as landfills and industry. Urban influences are believed to account for almost 60% of wetlands loss in the United States (Opheim, 1997).

In the northeastern U.S., the remaining urban tidal marshes are frequently dominated by *Phragmites australis* (common reed), a plant whose presence typically indicates an altered and degraded habitat (Winogrond & Kiviat, 1997; Chambers, McComb & Tappeiner, 1999). Few studies have focused on avian use of *Phragmites*-dominated marshes; however, it is generally thought that these degraded marshes provide limited resources for wetland birds (Roman, Niering & Warren, 1984; Benoit & Askins, 1999). The physical structure of dense, monospecific stands of *Phragmites* prevents shorebirds, waders, waterfowl, and other taxa from gaining access to the marsh surface for foraging.

The use of tidal-marsh restoration has increased in response to the continued degradation of wetlands and an enhanced understanding of the value of wetlands in urban areas (Zedler, 1996; Bergen, Alderson, Bergfors, Aquila & Matsil, 2000; Harbor Estuary Program [HEP], 2001). However, urban marshes have typically experienced a long history of perturbation and continue to be influenced by urban pressures. Under these circumstances, restoration potential is compromised, and evaluation of restorations based on comparison to conditions at reference sites may not be possible or appropriate. Instead, restorations can be evaluated using a same-site, "pre-restoration versus post-restoration" monitoring strategy. Use of this type of monitoring in tidal-marsh restoration has been rare, and thus it is an important focus for study.

Much literature exists demonstrating the importance of tidal-marsh habitat to numerous bird species throughout the year (Chavez-Ramirez & Slack, 1995; Reinert & Mello, 1995). However, relatively few studies have focused on avian response to marsh restoration, and of these, only a small number have taken place in urban areas. This is despite the fact that in terms of restoration monitoring, birds may be particularly good indicators of habitat quality because they integrate multiple environmental influences in a habitat and respond quickly to changes in habitat (Neckles et al., 2002; U.S. Environmental Protection Agency [EPA], 2002).

The goal of this study was to evaluate avian response to tidal-marsh restoration in an urban landscape. The objectives were to 1) compare and contrast the pre-restoration and post-restoration avian community in a restored marsh; and 2) to compare temporal, post-restoration trends in the avian community at two restored tidal marshes in close proximity to one another.

**Study Sites**

The New Jersey Hackensack Meadowlands (the Meadowlands) is a mosaic of brackish and freshwater tidal wetlands, freshwater non-tidal wetlands, uplands, and developed areas that includes the largest remaining tidal-marsh complex (3,400 hectares) in the New York–New Jersey Harbor Estuary (Figure 1a). Seven miles west of New York City, the Meadowlands lies along the Atlantic Flyway and is surrounded by an urban matrix. Over 90% of estuarine marshes in the Meadowlands are dominated by *Phragmites* due to decades of land alteration that created conditions favoring invasion by this species (Sipple, 1972; Tiner, Swords & McClain, 2002). Two brackish marshes in the Meadowlands—Harrier Meadow and Mill Creek—were included in this study (Figure 1b).

Harrier Meadow is a 32.2-hectare tidal marsh surrounded by tidal mudflats on two sides and urban development and landfill on the remaining two sides (Figure 2). Prior to restoration, Harrier Meadow was dominated by *Phragmites* and *Lythrum salicaria* (purple loosestrife) and featured scattered, small pools of water and isolated patches of native high-salt-marsh vegetation, such as *Spartina patens* (saltmeadow cordgrass) and *Distichlis spicata* (saltgrass). Just under 22 hectares (71%) of Harrier Meadow were restored in 1998. The restoration design included the creation of three large, tidally influenced open-water areas surrounded by high-marsh and fringe-upland vegetation. The remaining area, which consisted of high-marsh vegetation and *Phragmites*, was not altered. Restoration of Harrier Meadow was intended to create and enhance a variety of habitats for wildlife and to bring about the recovery of wetland function (Hartman, 2002a).

Mill Creek is a 56.7-hectare tidal marsh bordered by the New Jersey Turnpike on the east and residential land use on the west (Figure 3). Prior to restoration, Mill Creek was dominated by *Phragmites* and contained very little open water. General restoration goals for Mill Creek Marsh were similar to those for Harrier Meadow (Hartman, 2002b). Thirty-eight hectares (67%) of Mill Creek were restored in 1999, creating a tidal channel, tidally influenced open-water areas, and mudflats with interspersed islands of upland vegetation.

Approximately the same percentages of the two marshes were restored; however, Harrier Meadow and Mill Creek had distinctly different habitat compositions after restoration (Table 1a). Almost a third of Harrier Meadow (mostly in the unrestored portion of the marsh) was still covered by *Phragmites*, while this species was a negligible habitat component of Mill Creek. Mudflat, open water, upland, high marsh (Harrier Meadow only), and low marsh (Mill Creek only) habitats were significant components of both marshes, but in different configurations. These differences represent one of the challenges in evaluating responses to restoration, in that there is rarely an opportunity to have true replicates in restoration design. In this study, comparison of two designs enabled us to evaluate the general avian response to marsh restoration because both marshes were located within the same habitat complex and drew avian species from the same regional species pool. We were also able to contrast how variation in habitat availability led to differences in avian response.

## Methods

### Habitat Analysis

Digital habitat maps of Harrier Meadow and Mill Creek were created with geographic information system (GIS) software, Environmental Systems Research, Inc.'s ArcInfo 9, using color infrared digital aerial photographs and detailed field observations. Area coverage of habitat types was calculated in ArcMap using the digital vegetation maps. Habitats at Harrier Meadow included high marsh, mudflat, open water, *Phragmites*, and scrub-shrub/upland (Table 1b). Habitats at Mill Creek included low marsh, mudflat, open water, *Phragmites*, and upland.

### Avian Surveys

We conducted pre-restoration surveys of birds at Harrier Meadow in 1997. Mill Creek was not surveyed prior to its restoration due to the dense stands of *Phragmites* at the site, which severely limited the ability to conduct surveys. Post-restoration surveys were conducted at both marshes in 2001, 2002, and 2003. In all survey years, we surveyed each marsh five times during each of three seasons: spring migration (mid-March through mid-May), summer (early June through late July), and fall migration (early August through mid-October). Each marsh was therefore surveyed 15 times per survey year.

We conducted the avian surveys by scanning predetermined, fixed areas (stations) within each marsh and recording all individuals detected visually and audibly within a station during a five-minute period (see Seigel, 2006, for further explanation). Survey stations were delineated prior to restoration and did not change during monitoring. One avian survey consisted of a count at all stations present at the marsh. There were six survey stations at Harrier Meadow (four within the restored portion of the marsh and two in the unrestored portion) covering a total of 9.8 hectares. Included in the total survey area were 2.9 hectares located outside the marsh boundaries in the surrounding mudflat. Mill Creek contained five survey stations (all within the restored portion of the marsh) covering 14 hectares. Individual birds flying over the marsh were not recorded, with the exception of foraging raptors and aerial insectivores. Care was taken not to count the same individual twice. We surveyed the marshes between sunrise and 10 a.m. Surveys were not conducted in rain or heavy wind. Species such as bitterns and rails were underrepresented by the surveys because we did not use vocalization tapes for these species.

Individual birds were identified to the species level. In three instances, however, individuals were identified to the genus level due to an inability to distinguish between species (American crow and fish crow, recorded as *Corvus* species) or difficulty in identifying individuals in large flocks (greater and lesser yellowlegs, recorded as *Tringa* species, and semipalmated and least sandpipers, recorded as *Calidris* species).

### Analysis

We assessed avian response to restoration using two approaches. The first compared the pre- and post-restoration avian communities at Harrier Meadow. To keep sampling effort equal, we compared one year of pre-restoration data (1997) with one year of post-restoration data (2002), the middle year of the three-year post-restoration data set. Our second approach examined post-restoration (2001–2003) trends in the avian communities at both of the restored marshes.

Metrics used to characterize the avian communities included avian abundance, species richness, diversity, evenness, and Sorensen's similarity.

We also studied foraging guilds to examine how the change in habitat structure at each marsh affected the avian communities. The avian community was categorized into six guilds: generalist, aerial, upland, *Phragmites*, open-water, and mudflat foragers (Table 2). Generalists included species that foraged in more than one type of habitat. Species were placed in a guild based on the predominant habitat used for foraging (based on Ehrlich, Dobkin & Wheye, 1988; personal observation). There was only one species (common snipe, *Gallinago gallinago*) that was observed foraging in high-marsh habitat, and therefore this habitat and associated species were not included.

We realize that our method of analysis does not contain true replication in avian response to restoration. However, with the exception of extremely well funded projects, replication is nearly impossible in this type of study. In any case, repeated, independent studies of numerous restorations are necessary for developing strong inferences regarding patterns and cause-effect relationships in avian response to restoration. While we understand the limitations of our approach for statistical analysis, the careful documentation this study represents is an essential first step toward developing a body of scientific research.

We took a very conservative approach to analysis by using descriptive statistics (mean ± standard error) to detect differences in avian community characteristics (abundance, species richness, diversity, and evenness) before and after restoration at Harrier Meadow, and between the restored Harrier Meadow and restored Mill Creek. Diversity of the avian communities was measured using the Shannon-Wiener diversity index (H'): $H' = -S(p_i) \times (\ln p_i)$, where $p_i$ is the proportional abundance of species $i$, summed for all $n$ species measured. Evenness is defined as the Shannon diversity divided by the maximum possible diversity (Krebs, 1989). Similarity of species composition of avian communities between two years was determined using Sorensen's similarity index ($C_s$): $C_s = 2j / (a+b)$, where $j$ is the number of species present in both years, $a$ is the number of species in the first year, and $b$ is the number of species in the second year (Magurran, 1988).

Because total survey areas at Harrier Meadow and Mill Creek were not equal, comparisons of avian community characteristics between the two marshes were standardized by factoring in the number of hectares surveyed. We divided average avian abundance by the number of hectares surveyed to obtain average density per survey. Species richness was also divided by the number of hectares surveyed and is presented as the number of species observed per hectare per survey. The species-richness-per-hectare measure should be considered with caution, however, as the relationship between species richness and area isn't necessarily linear.

## Results

### Pre-restoration Versus Post-restoration at Harrier Meadow

Through the addition of open water and other habitats during restoration, Shannon diversity of habitat types at Harrier Meadow increased from 0.965 in 1997 to 1.121 in 2002. A total of 43 bird species were observed in the marsh in 1997. In comparison, cumulative species richness in 2002 was 57, an increase of more than 30% after restoration.

Similarity of species composition in the avian community before and after restoration was 0.62.

Average avian abundance per survey exhibited more than a tenfold increase, from 33 (± 3) prior to restoration to 453.80 (±184) after restoration (Figure 4). Red-winged blackbird (*Agelaius phoeniceus*) and sandpipers (*Calidris* species) were the most abundant species before and after restoration, respectively. Average avian species diversity per survey was essentially the same before and after restoration (Table 3). Changes to the elements of diversity—species richness and evenness—were more indicative of the influence of restoration. On a per-survey basis, pre-restoration species richness was approximately half that of post-restoration. Evenness per survey decreased by 24% after restoration.

To determine if the large flocks that periodically visited the marsh masked a change in diversity after restoration, we removed sandpipers (*Calidris* species) from the analysis. Without the considerable reduction in evenness caused by the temporal variability of large sandpiper flocks, annual diversity was higher (2.11 ± 0.08) after restoration.

To examine how changes in habitat types after restoration influenced avian community structure, we compared the average abundance per survey of foraging guilds before and after restoration (see Table 4 for standard errors). Prior to restoration, all guilds were present in low abundance except for aerial foragers, which were absent (Figure 5). The dominant foraging guild was generalists. After restoration, average abundance per survey increased markedly in three of the six guilds: Mudflat foragers increased from 0.73 (± 0.05) to 294 (± 189), open-water from 5.5 (± 1.2) to 84.9 (± 12.4), and generalists from 14.9 (± 2.3) to 58.9 (± 18.7). The only guild to decrease in abundance after restoration was *Phragmites* foragers, which dropped from 5.6 (± 1.3) to 3.3 (± 0.8) individuals per survey.

Species richness within foraging guilds showed a pattern of post-restoration change generally similar to that seen in abundance (Table 5). Mudflat and open-water foragers showed the largest percent increase in species richness after restoration. Although three additional species were present after restoration in the upland foraging guild, seven species were absent, resulting in a net loss of species. Pre- and post-restoration within-guild species composition was similar for generalist, upland, and open-water foragers.

The marshes of the Meadowlands serve as important stopover sites during spring and fall migration and breeding habitat in the summer. Seasonal changes in avian community composition are masked when examining the effect of restoration only on the annual level. Therefore, we examined pre- and post-restoration seasonal patterns in the avian community. Prior to restoration, average avian species richness per survey did not vary a great deal by season, though there was a trend for species richness to decrease from spring to fall (Figure 6). In contrast, average species richness per survey increased considerably after restoration in the spring, summer, and fall. After restoration, fall species richness was lower than spring and summer because there were 22 species, mostly waterfowl and passerines, present in the marsh in spring and/or summer but not in fall. Before restoration, nine of 43 species were observed in all three seasons, and three species (0.07%) had a frequency = 0.67. After restoration, 27 of 57 species were observed in all three seasons, while 12 species (21%) had a frequency = 0.67 (Table 6).

Average diversity per survey was similar across seasons prior to restoration (Table 7). In contrast, post-restoration diversity decreased from spring to fall, and there was a tendency for greater variability in diversity in summer and fall. Pre- and post-restoration seasonal diversity were similar. Post-restoration spring had the highest diversity, primarily due to a doubling in species richness from pre-restoration accompanied by a relatively small decrease in evenness. Diversity in the fall decreased due to a relatively large decrease in evenness. The highest degree of seasonal variability in diversity accompanied the decrease in diversity in the fall.

Prior to restoration, guild structure was relatively constant across all seasons, with generalists most abundant in spring and summer and decreasing in fall (Figure 7). There were more birds in each of the six guilds after restoration, with a marked increase in generalist abundance in spring and summer and the addition of mudflat and open-water foragers across all seasons. Post-restoration spring and summer guild structure was somewhat similar, whereas fall abundances were heavily dominated by mudflat foragers. Abundance of open-water foragers remained somewhat similar across the three seasons, whereas generalists declined in the fall.

### Post-restoration Avian Trends at Harrier Meadow and Mill Creek

Shannon diversity of habitat types at Harrier Meadow (1.12) was only slightly higher than at Mill Creek (1.09), despite the considerable difference in habitat configuration (Figures 2 and 3; Tables 1a and 1b).

Across the three years surveyed after restoration, cumulative species richness was 78 species (8 species per hectare) at Harrier Meadow and 65 species (4.6 species per hectare) at Mill Creek. Similarly, on a per-survey basis, Harrier Meadow also had a higher average number of species per hectare than Mill Creek in all years after restoration (Table 8). Within each individual marsh, species richness per hectare did not vary across post-restoration years. Density was also consistently higher at Harrier Meadow than at Mill Creek after restoration (Figure 8). Both marshes exhibited a decrease in avian density over time; however, the decrease was substantial only at Harrier Meadow, with density decreasing by 60% between 2001 and 2003. Though there was considerable variability, the largest consecutive interannual decline (59%) occurred between 2002 and 2003 at Mill Creek. At both marshes, the five species with the highest densities were Canada goose (*Branta canadensis*), mallard (*Anas platyrhynchos*), snowy egret (*Egretta thula*), great egret (*Casmerodius albus*), and gadwall (*Anas strepera*).

After restoration, avian diversity and evenness did not differ markedly between marshes in any year or among years at either marsh (Table 8).

Both Harrier Meadow and Mill Creek were dominated by three foraging guilds: open-water, mudflat, and generalist. While the relative proportions of guild density were very similar in 2001 at both marshes (Figure 9), the density of generalists, open-water, mudflat, and upland foragers at Harrier Meadow was more than double the density at Mill Creek (Figure 10). Average density of generalists decreased significantly between 2001 (10.6 ± 1.7) and 2003 (3.4 ± 0.7) at both Harrier Meadow and Mill Creek (2001: 4.6 ± 1.3; 2003: 0.89 ± 0.2). At Mill Creek, the density of open-water foragers significantly decreased between 2001 (6.2 ± 1.1) and 2003 (3.2 ± 0.7).

Migratory flocks of *Calidris* species sandpipers made a disproportionately large contribution to overall avian density at both Harrier Meadow and Mill Creek after restoration. In 2001 and 2002, sandpipers accounted for over 90% of mudflat-forager abundance and over 60% in 2003. Average flock size was similar at Harrier Meadow (488 ± 171) and Mill Creek (447 ± 184). The average flock size of sandpipers decreased at both marshes from 2001 (Harrier Meadow 874 ± 384; Mill Creek 809 ± 558) through 2003 (Harrier Meadow 191 ± 119; Mill Creek 108 ± 44).

## Discussion

Several aspects of the study design proved to be of particular value in assessing the avian response to restoration at Harrier Meadow. Most notably, the pre-restoration data for Harrier Meadow allowed us to make a direct, same-site comparison of the avian community present in the pre- and post-restoration habitats. The exclusive use of baseline or pre-restoration avian data from a separate marsh might have introduced confounding factors such as landscape context and hydrology and complicated the assessment of how restoration affected the avian community. Studying avian community structure at multiple temporal scales (annual, seasonal, and multiyear trends) also proved to be informative because we were able to detect patterns in the avian community typically hidden at the annual scale. Furthermore, the use of foraging guilds allowed us to gain a better understanding of how changes in the availability of particular resources in the marsh may influence avian habitat use (Miller & Cale, 2000).

### Pre-restoration Versus Post-restoration

Changes in avian community structure at Harrier Meadow reflected the change in habitat heterogeneity resulting from restoration. Prior to restoration at Harrier Meadow, breeding-season species richness was low (Burger, Shisler & Lesser, 1982), and abundance was dominated by a single species, the red-winged blackbird. This type of single-species-dominated community was consistent with avian communities found in other marshes with low habitat heterogeneity and a lack of surface water (Moller, 1975; Reinert, Golet & DeRagon, 1981; Craig & Beal, 1992).

Restoration of Harrier Meadow increased habitat heterogeneity considerably, including a large increase in open-water habitat. Avian community structure also changed considerably. The most prominent changes included a significant increase in avian abundance and an accompanying transition from a community dominated by passerines to one dominated by waterbirds. This type of avian response is supported by other studies that found a direct relation between the amount of surface water available in a marsh, avian abundance, and the use of the marsh by waterbirds such as waterfowl, shorebirds, and wading birds (Reinert et al., 1981; Burger et al., 1982; Slavin & Shisler, 1983).

The availability of open water at Harrier Meadow may have also affected the temporal dynamics of the avian community. After restoration, avian community composition was much less variable across consecutive surveys. The pre-restoration variability may have been caused by unpredictability of water availability in the marsh. With tidal influence restored and new ponds created, there was a greater area of surface water in the marsh, and the availability of water was more persistent throughout the year. As a result, avian species that relied on water for various activities, including foraging, were consistently present in the restored marsh, whereas they were occasional visitors in pre-restoration surveys.

### Temporal Avian Responses to Restoration

The density of open-water foragers at Harrier Meadow was more than double that at Mill Creek, despite the fact that Mill Creek contained a greater area of open water. This implies that factors other than habitat availability per se influenced avian use of open water in the marshes.

This study did not include a means of rigorously determining what these factors were; however, one possible explanation became apparent during the study, and it has implications for future restoration design: Greater habitat heterogeneity and the configuration of the habitat mosaic at Harrier Meadow may have been responsible for the greater density of open-water foragers observed there. *Phragmites* and high-marsh grasses grew along the perimeter of open-water areas and extended outward at Harrier Meadow, creating protective cover and resting habitat for wading birds, shorebirds, and waterfowl (Reinert & Mello, 1995).

At Mill Creek, however, the band of vegetation along the perimeter of open-water areas was narrow compared with that at Harrier Meadow, and there were no large expanses of high marsh. Mill Creek therefore had limited resting area and protective cover for birds, possibly leading to limited use of open water. Other factors not addressed in this study, such as water depth, food availability, adjacent land use, and wildlife management may also have contributed to differential use of the two marshes studied.

In the mudflat foraging guild, the average flock size of *Calidris* species sandpipers decreased at both marshes from 2001 through 2003. During the same time period, but at Harrier only, we also saw a decrease in green-winged teal (*Anas crecca*), another species that forages on the mudflat in both marshes during migration. Concurrently, vegetation became established on what were previously bare mudflats, a phenomenon that has been linked to a decrease in avian use (Eertman, Kornman, Stikvoort & Verbeek, 2002). We do not believe the vegetated areas were large enough to cause the observed decrease in sandpiper abundance and suggest this as an interesting focus for future research. Several species in other guilds also decreased in abundance during the post-restoration monitoring period. In the open-water foraging guild, for example, mallard decreased at both marshes while great and snowy egrets decreased at Mill Creek.

It is possible that the observed changes in abundance at Harrier and Mill Creek were actually occurring on a scale larger than the local level and were not a result of post-restoration habitat changes in the marsh. For example, the decreasing trend in abundance of *Calidris* species sandpipers at Harrier Meadow and Mill Creek may not have been due to the loss of mudflat habitat at the restoration sites, but rather to population trends occurring at the broader, regional scale. Two sources of data would be useful in determining the scale at which changes in the avian community occurred: 1) regional-scale population data for the avian species observed at the restored marshes, and 2) avian community data from spatially independent, stable marshes within the Meadowlands. (We use the term "stable" to refer to a marsh that has not experienced a major disturbance, such as restoration or change in hydrologic pattern, for more than a decade.) If trends in the avian community at the restoration sites were similar to trends found outside the marsh, a "restoration effect" may be ruled out. Conversely, if trends within each restored marsh were unique to that locality, they may be interpreted to be a result of ecological dynamics associated with succession after restoration.

It appears that the restorations at Harrier and Mill Creek were successful because each newly created habitat was associated with increased avian use. However, with such a broad restoration goal as "to increase avian use of the marsh," it is very easy to deem both restorations successful. A positive or negative evaluation of these restorations would be more meaningful if it were based on more clearly defined goals. For example, the abundance of open-water foragers at Mill Creek increased considerably after restoration. This aspect of the restoration would be judged a success under the broad goal of increasing avian use of the marsh. If we take a closer look, however, we see that mallard accounted for 60% of the increase. If the goal were more specific in terms of target species, the general increase in open-water foragers might not be considered as successful. We stand to learn more about the restoration process by studying the success or failure to achieve clearly defined goals and realistic goals (Kentula, 2000; Choi, 2004).

In this study it was not possible to locate true reference sites for comparison with Harrier Meadow and Mill Creek. The urban context of the marshes placed numerous constraints on the restoration itself and on the evaluation of post-restoration progress (Grayson, Chapman & Underwood, 1999; Ehrenfeld, 2000). For example, the restoration plans had to take into consideration limitations due to habitat fragmentation and tidal restriction. Consequently, the marshes were not restored to the habitat composition typical of a salt marsh or any other naturally preexisting habitat type, hampering the use of reference sites. Moreover, the majority of marshes in the Meadowlands could not serve as reference sites as they are themselves ecologically degraded and dominated by *Phragmites*.

Few studies of avian response to marsh restoration in urban landscapes exist in the literature. This study serves as an essential first step toward developing a body of scientific research addressing avian response within the unique urban context. We suggest two particular areas in which future studies should be focused. First, our avian survey data showed differences between the two marshes in abundances of each foraging guild. Foraging studies directly linking birds and specific restored habitat types would provide insight into the mechanisms behind the success and failures in achieving restoration goals. Second, future studies should also investigate reproductive success in restored marshes in urban habitats. It is important to determine whether these urban marshes, which are under constant anthropogenic pressure (Ehrenfeld, 2000), serve as sources or sinks for the birds breeding on site.

While post-restoration monitoring has provided insights into the avian community response to marsh restoration, it is important to note that three years of post-restoration monitoring is a relatively short period from which to draw conclusions about the long-term trajectory of the marshes. As more long-term monitoring data become available, we are learning that it could take one to several decades for restored wetlands to recover ecological function, and that the time frame for a restoration is dependent upon the initial conditions of the wetland (Zedler & Callaway, 1999; Warren et al., 2002).

## Acknowledgments

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# Glossary

**Anthropogenic:** Caused by humans.

**Descriptive statistics:** Statistics used to describe the basic features of the data in a study, as distinct from inferential statistics, which attempt to reach conclusions that extend beyond the immediate data alone. (For more information, see the Center for Social Research Methods web sites at http://www.socialresearchmethods.net/kb/statdesc.htm.)

**Diversity:** A metric frequently used to describe a community based on species richness and the relative abundance of each species.

**Evenness:** A measure of how similar the abundances of species are within a habitat. When there are similar proportions of all species, then evenness is near one, but when the abundances are very dissimilar (i.e., some rare and some common species), the value approaches zero.

**Generalist:** Adapted to a broad range of habitats.

GIS (geographic information system): A computer system capable of integrating, storing, editing, analyzing, and displaying geographically referenced information.

**Guild:** A group of species, all members of which exploit similar resources in a similar fashion (Oxford Dictionary of Ecology).

**Metric:** A standard of measurement for estimating or indicating a specific characteristic or process.

**Monospecific:** Single species.

**Reference site:** A model ecosystem used for planning and evaluating an ecological restoration project. Typically, the reference represents a point of advanced development that lies somewhere along the intended trajectory of the restoration. (The SER International Primer on Ecological Restorations; see www.ser.org.)

**Sink:** Less-suitable habitat that can act as a reservoir for surplus populations migrating from more suitable (“source”) habitat. In sink habitat, mortality exceeds reproduction.

**Sorensen’s similarity index:** An index that compares species presence and absence between habitats. Values approaching zero indicate lower similarity between the communities; values approaching one indicate higher similarity.

**Species richness:** The number of different species found in a particular habitat.

**Succession:** The sequential change in vegetation and the animals associated with it, either in response to an environmental change or induced by the intrinsic properties of the organisms themselves.

**Figure 1a. Map of the New Jersey Hackensack Meadowlands District. Boundaries are indicated by white lines.**

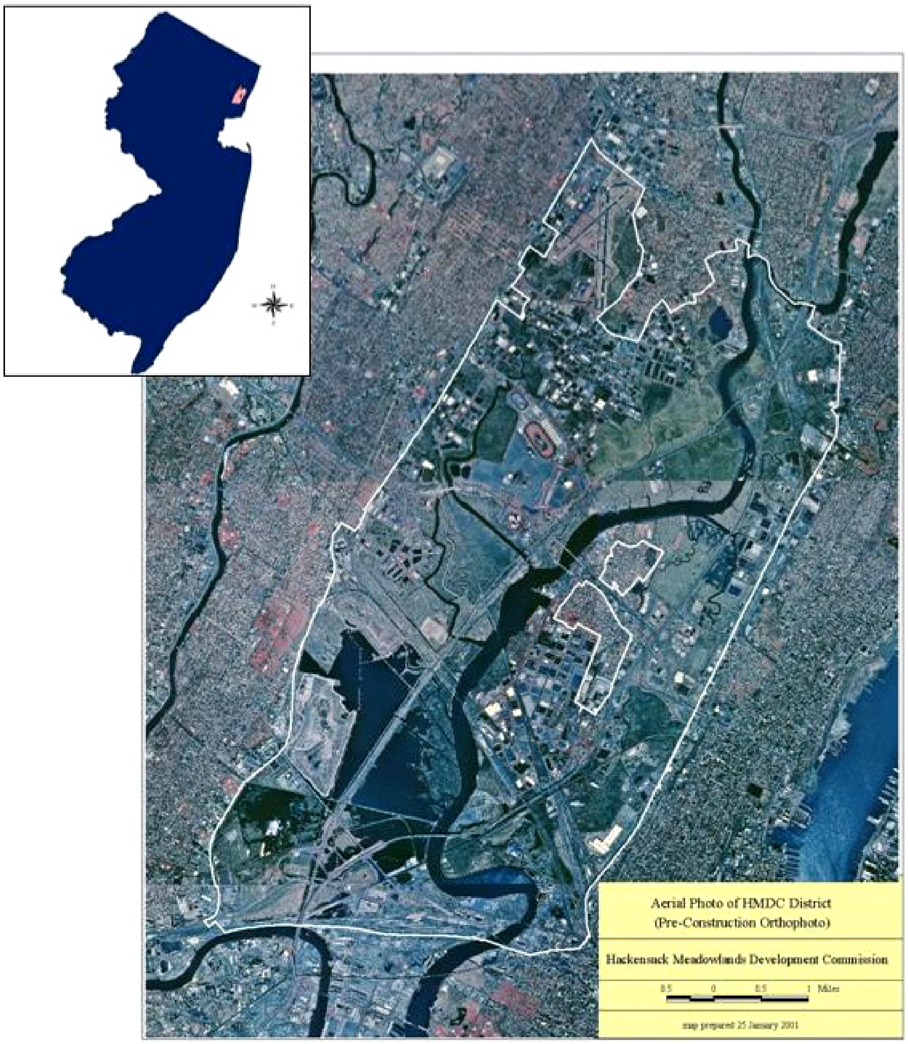

**Figure 1b. Location of Harrier Meadow and Mill Creek marshes within the Meadowlands District.**

**Figure 2. Distribution of six habitat types at Harrier Meadow Marsh after restoration. Though not illustrated here, the southeastern and southwestern sides of the marsh were adjacent to mudflats.**

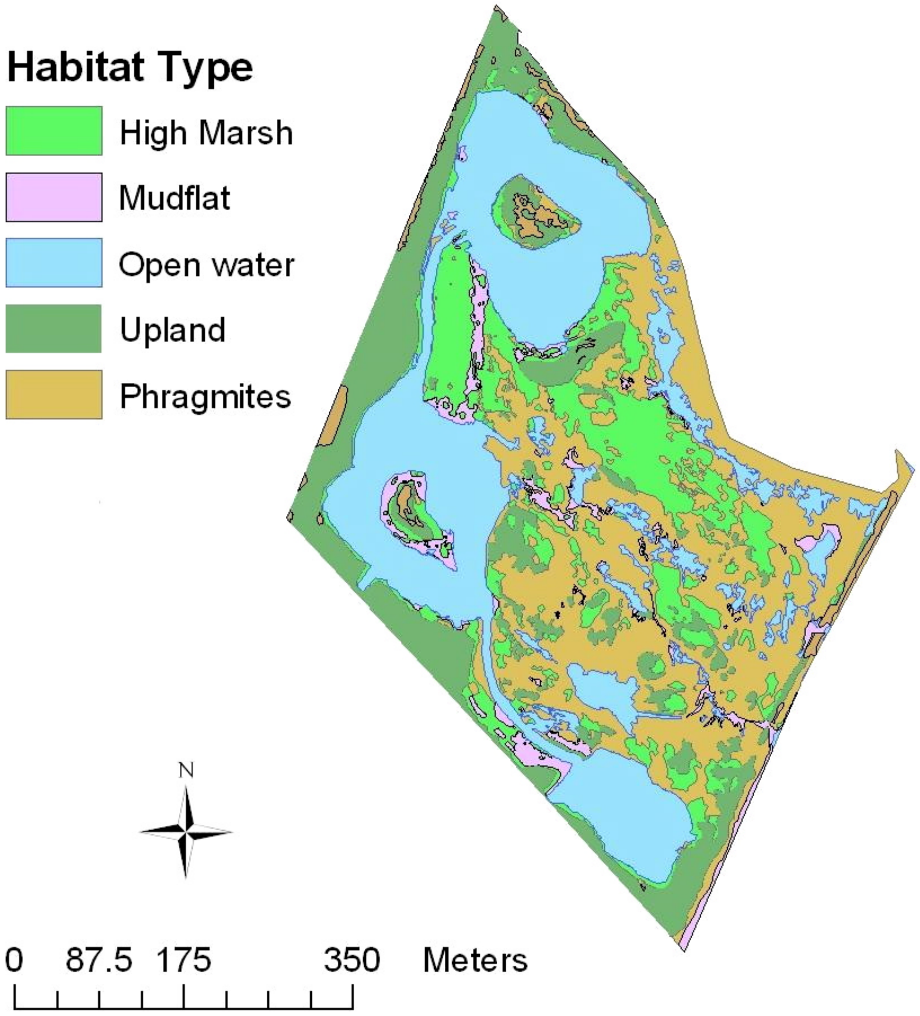

**Figure 3. Distribution of four habitat types at Mill Creek Marsh after restoration.**

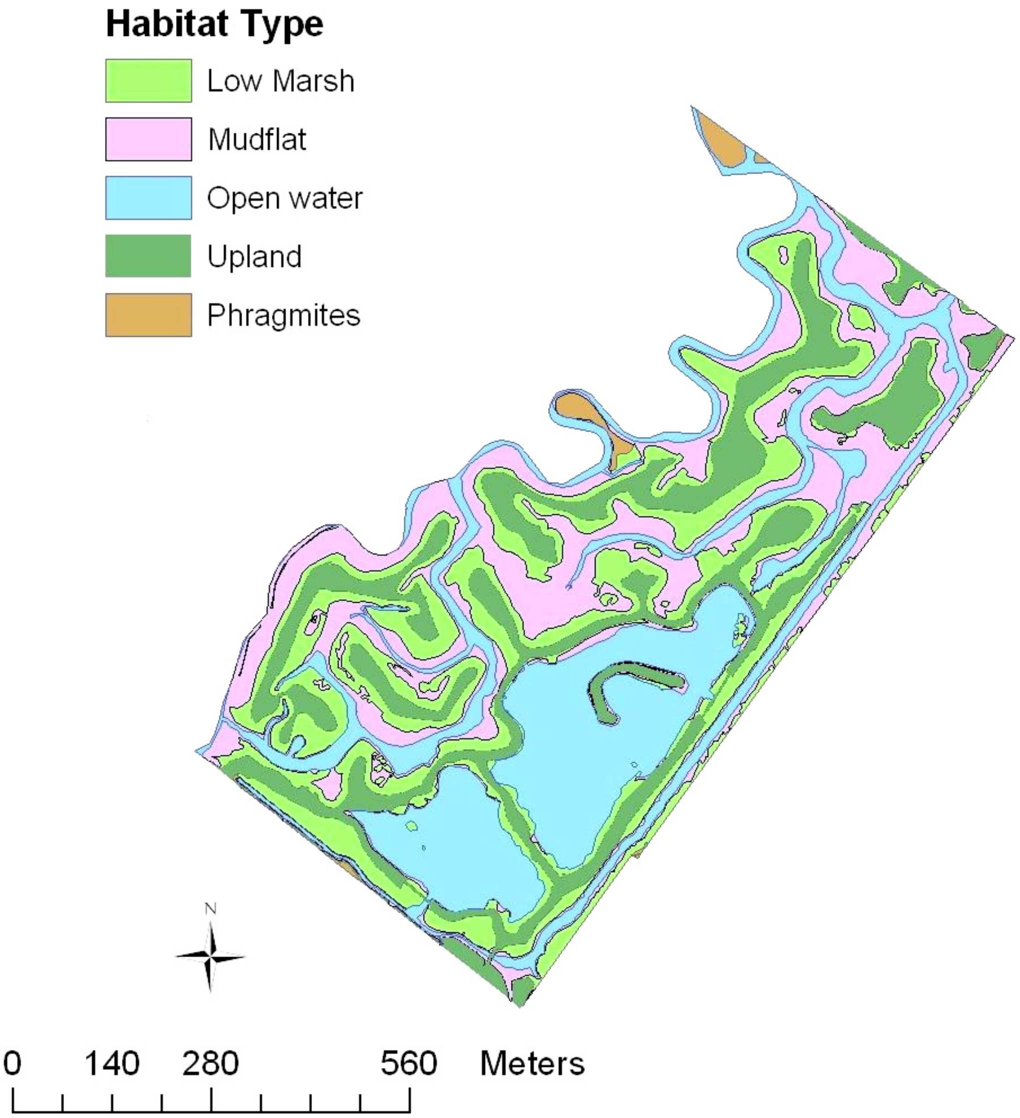

**Figure 4. Average (± SE) avian abundance per survey at Harrier Meadow prior to restoration (1997) and after restoration (2002).**

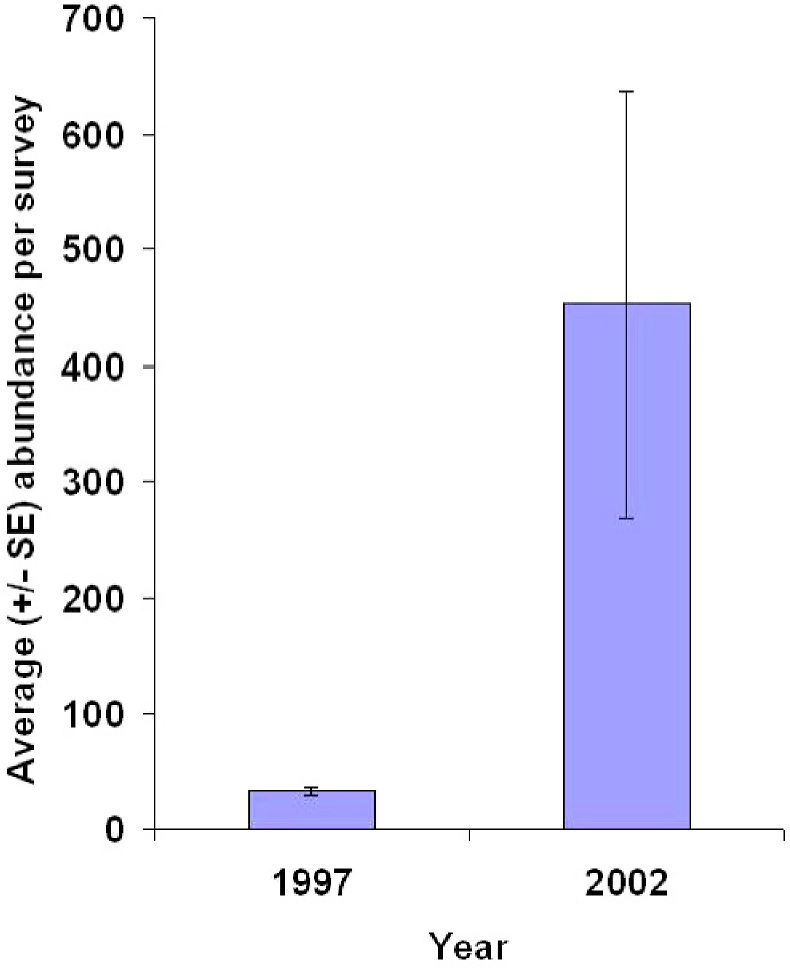

**Figure 5. Average abundance per survey of six foraging guilds before (1997) and after (2002) restoration at Harrier Meadow. Guilds include upland foragers (U), *Phragmites australis* foragers (P), open-water foragers (OW), mudflat foragers (MF), generalists (G), and aerial foragers (A).**

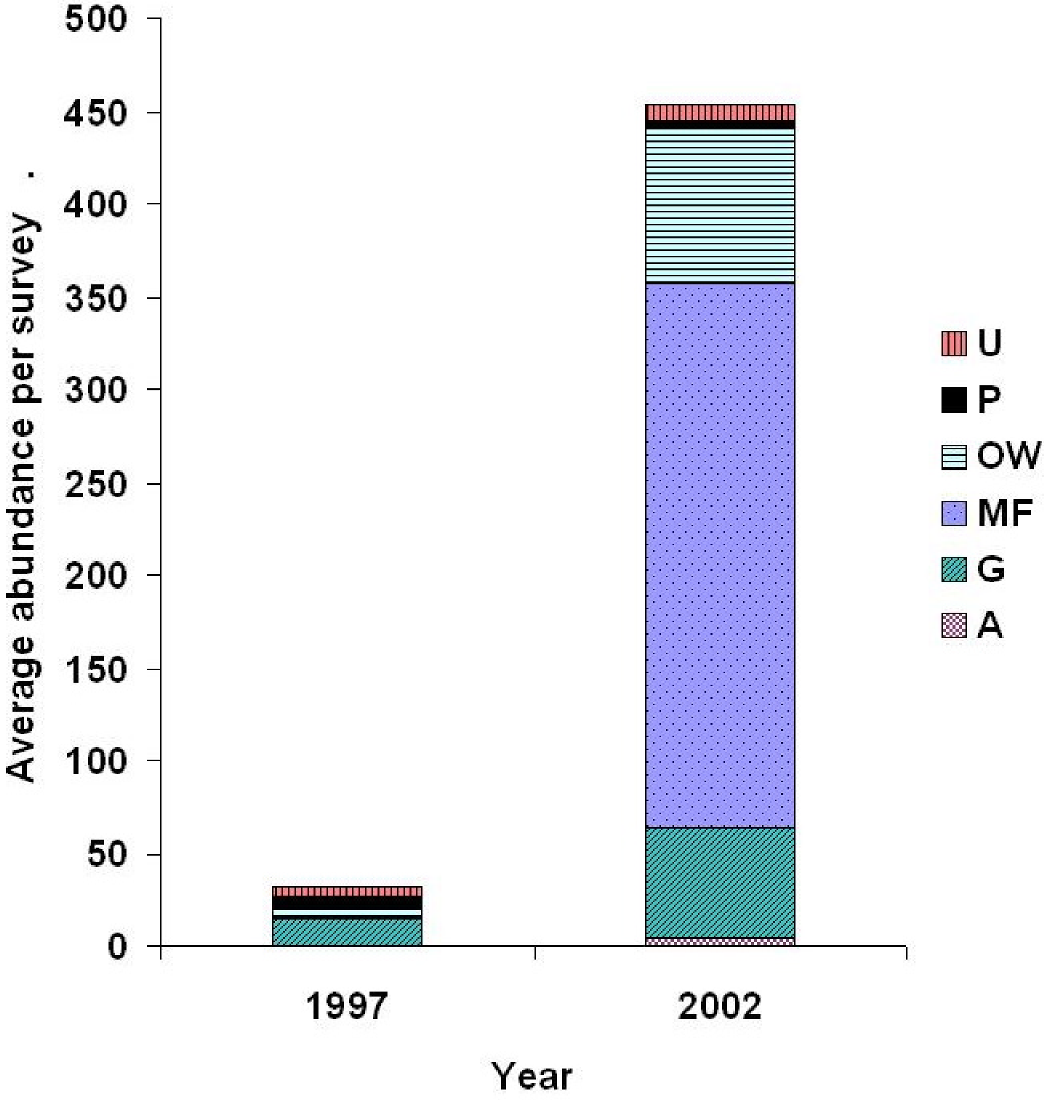

**Figure 6. Average (± SE) avian species richness per survey before (1997) and after (2002) restoration at Harrier Meadow during fall and spring migration and summer breeding season.**

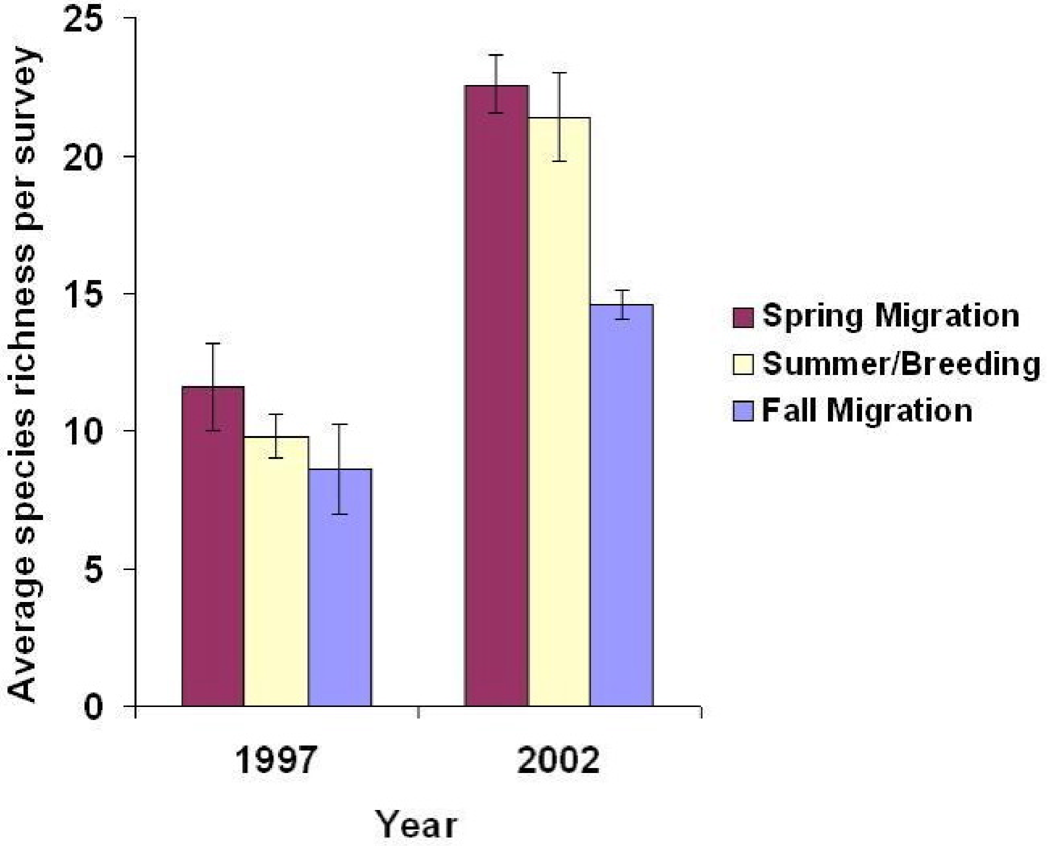

**Figure 7. Average abundance per survey of six foraging guilds at Harrier Meadow before (1997) and after (2002) restoration during spring, summer, and fall. Guilds include upland foragers (U), *Phragmites australis* foragers (P), open-water foragers (OW), mudflat foragers (MF), generalists (G), and aerial foragers (A).**

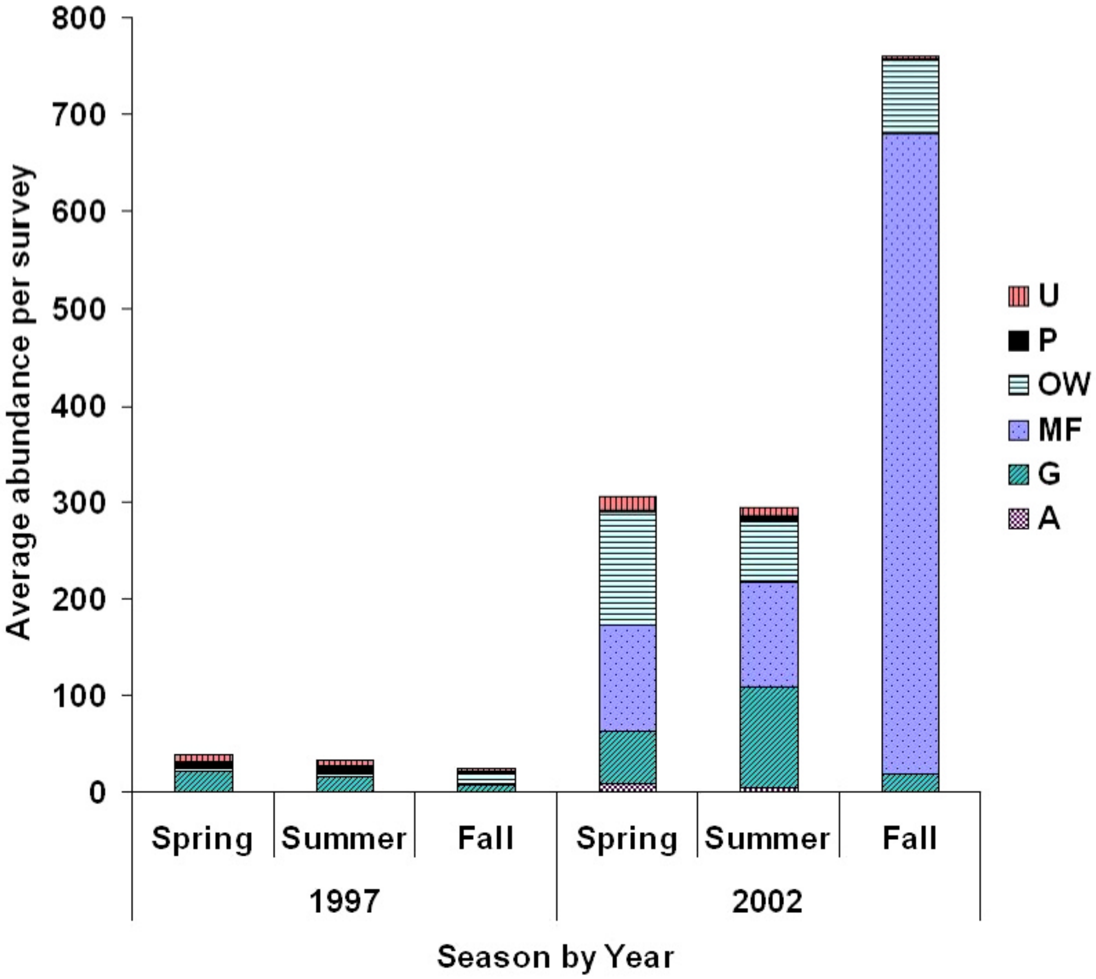

**Figure 8. Average (± SE) avian density per survey at Harrier Meadow and Mill Creek during three consecutive years after restoration.**

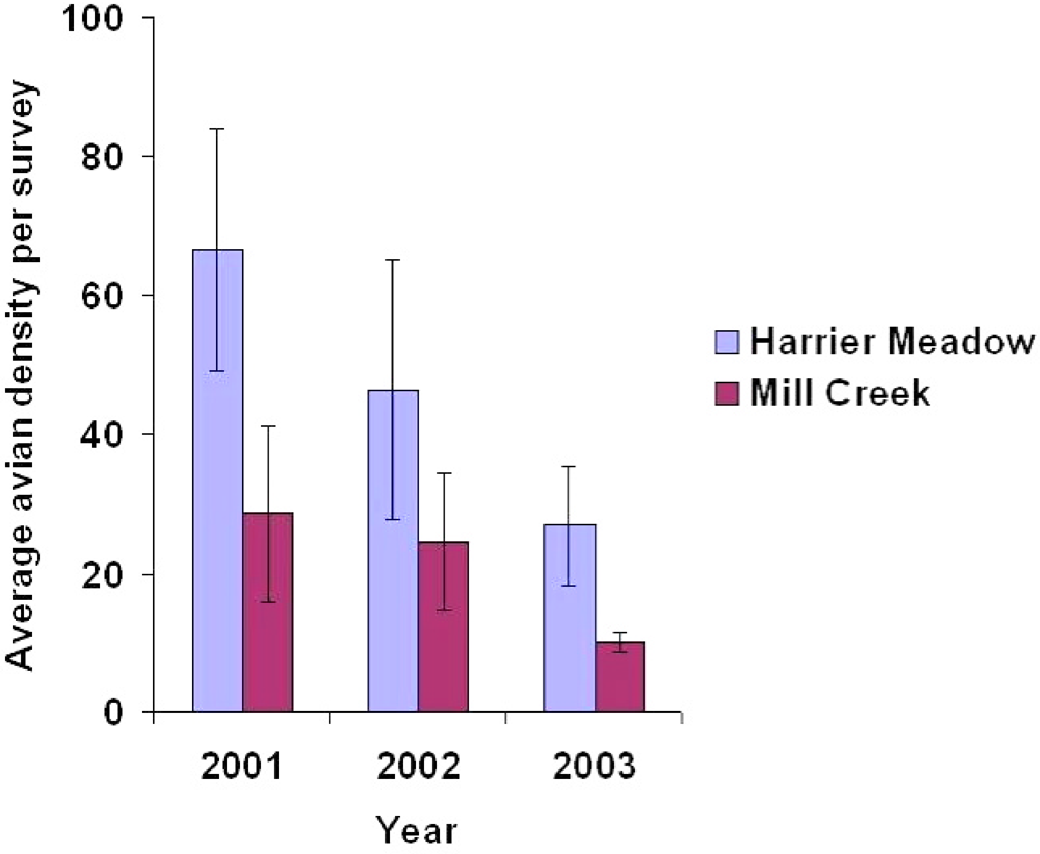

**Figure 9. Relative density of six foraging guilds at Harrier Meadow and Mill Creek after restoration. Guilds include upland foragers (U), *Phragmites australis* foragers (P), open-water foragers (OW), mudflat foragers (MF), generalists (G), and aerial foragers (A).**

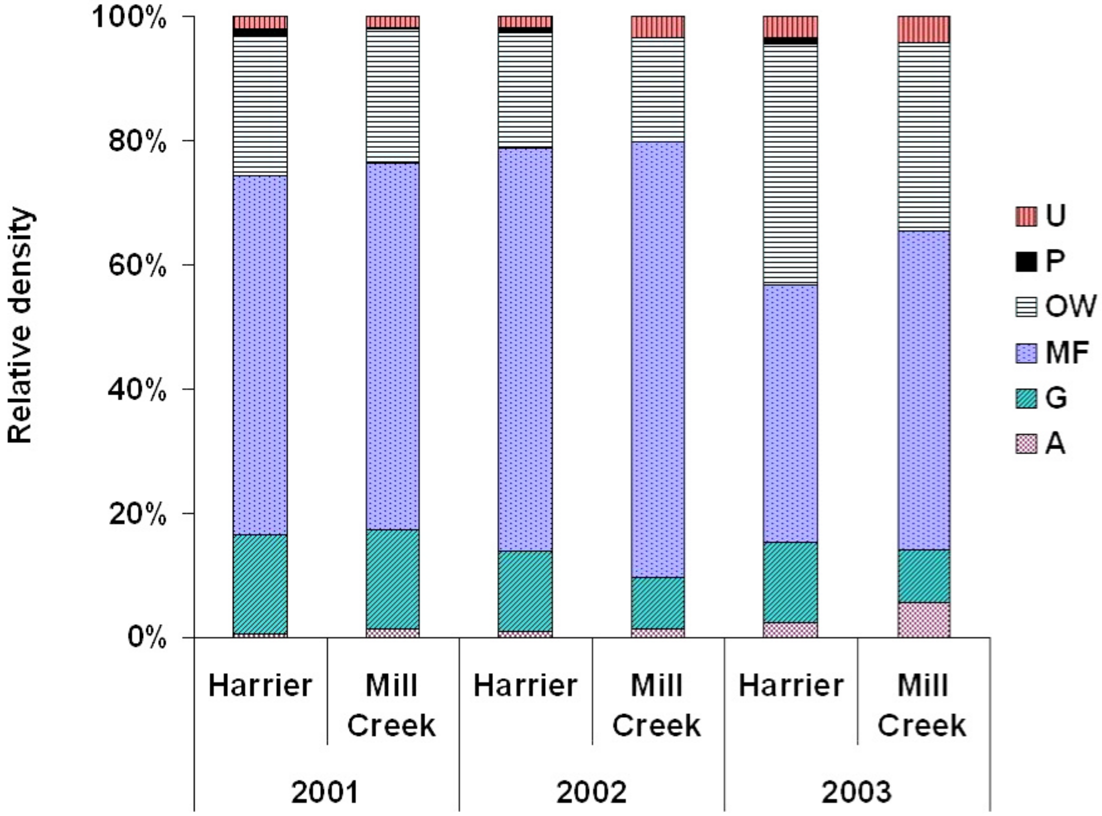

**Figure 10. Average density per survey for six foraging guilds at Harrier Meadow and Mill Creek after restoration. Guilds include upland foragers (U), *Phragmites australis* foragers (P), open-water foragers (OW), mudflat foragers (MF), generalists (G), and aerial foragers (A).**

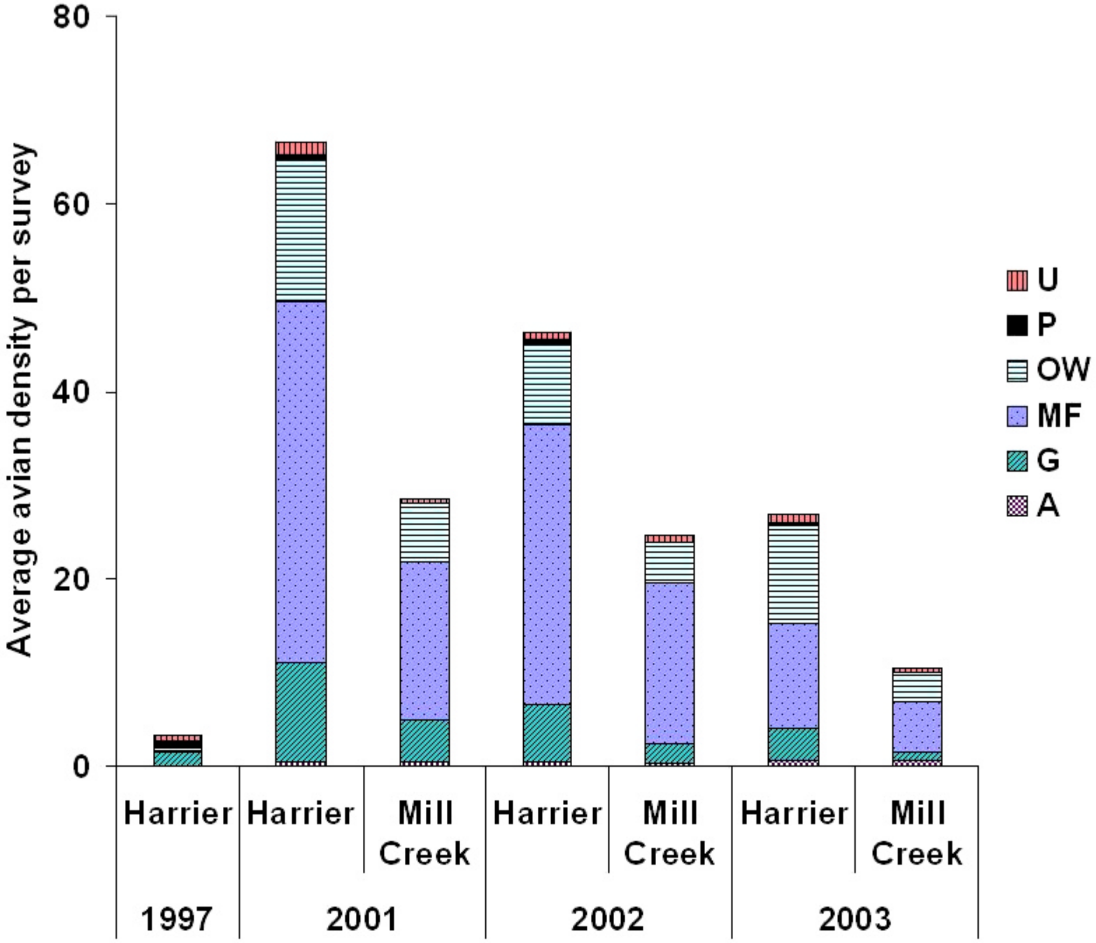

**Table 1a. Pre-restoration (1997) and post-restoration (2002) area coverage (in hectares) of six habitat types at Harrier Meadow and Mill Creek (post-restoration only). Habitat areas are presented for the avian survey area and for the entire marsh.**

| | **Harrier Meadow** | | | | **Mill Creek** | |
|---|---|---|---|---|---|---|
| | Pre-restoration | | Post-restoration | | Post-restoration | |
| | survey area | entire marsh | survey area | entire marsh | Survey area | Entire marsh |
| High marsh | 0.50 | 2.50 | 0.10 | 4.71 | 0 | 0 |
| Low marsh | 0 | 0 | 0 | 0 | 2.42 | 11.75 |
| Mudflat | 2.90 | 2.90 | 5.15 | 5.28 | 2.74 | 16.91 |
| Open water | 0 | 1.50 | 3.40 | 7.63 | 8.14 | 16.95 |
| Upland | 0.90 | 2.00 | 0.92 | 4.53 | 0.69 | 10.38 |
| *Phragmites* | 5.40 | 23.30 | 0.23 | 10.05 | 0.01 | 0.71 |
| **Total** | 9.80 | 32.20 | 9.8 | 32.20 | 14.00 | 56.70 |

**Table 1b. Relative proportion and actual area (in hectares) of six habitat types at Harrier Meadow and Mill Creek after restoration.**

| | **Proportion of entire marsh** | | **Entire marsh** | |
|---|---|---|---|---|
| | Harrier Meadow | Mill Creek | Harrier Meadow | Mill Creek |
| High marsh | 0.15 | 0.00 | 4.71 | 0 |
| Low marsh | 0.00 | 0.21 | 0 | 11.75 |
| Mudflat | 0.16 | 0.30 | 5.28 | 16.91 |
| Open water | 0.24 | 0.30 | 7.63 | 16.95 |
| Upland | 0.14 | 0.18 | 4.54 | 10.38 |
| *Phragmites* | 0.31 | 0.01 | 10.04 | 0.71 |
| **Total** | 1.0 | 1.0 | 32.30 | 56.7 |

**Table 2. Species composition of six foraging guilds. Species were placed in guilds based on Ehrlich et al. (1988) and personal observation.**

**Aerial**
Barn swallow
Bank swallow
Chimney swift
Eastern phoebe
Northern rough-winged swallow
Tree swallow

**Generalist**
Blue jay
Common grackle
*Corvus* species
European starling
Great black-backed gull
Herring gull
Laughing gull
Northern harrier
Northern waterthrush
Peregrine falcon
Ring-billed gull
Rock dove
Red-tailed hawk
Red-winged blackbird

**Mudflat**
Black-bellied plover
*Calidris* species
Dunlin
Green-winged teal
Killdeer
Semipalmated plover
Spotted sandpiper

***Phragmites***
Marsh wren
Sora
Swamp sparrow
Virginia rail

**Open water**
American black duck
American coot
American wigeon
Black-crowned night heron
Belted kingfisher
Black skimmer
Brant
Blue-winged teal
Canada goose
Common merganser
Common moorhen
Common tern
Double-crested cormorant
Forster's tern
Gadwall
Great blue heron
Great egret
Green heron
Hooded merganser
Mallard
Mute swan
Northern pintail
Northern shoveler
Osprey
Ring-necked duck
Ruddy duck
Snowy egret
Snow goose
Tri-colored heron
*Tringa* species
Wilson's phalarope
Wood duck

**Upland**
American goldfinch
American robin
American tree sparrow
Baltimore oriole
Black-capped chickadee
Brown-headed cowbird
Brown thrasher
Cedar waxwing
Common yellowthroat
Dark-eyed junco
Downy woodpecker
Eastern towhee
Gray catbird
House sparrow
Indigo bunting
Mourning dove
Northern cardinal
Northern flicker
Northern mockingbird
Palm warbler
Ruby-crowned kinglet
Ring-necked pheasant
Savannah sparrow
Song sparrow
White-crowned sparrow
Willow flycatcher
White-throated sparrow
Yellow warbler
Yellow-rumped warbler

**Table 3. Average avian diversity, evenness, and richness per survey at Harrier Meadow before (1997) and after (2002) restoration. Measurements are reported in mean ± standard error.**

| Year | Diversity | Evenness | Species Richness |
|---|---|---|---|
| 1997 | 1.80 ± 0.08 | 0.80 ± 0.02 | 10.00 ± 0.82 |
| 2002 | 1.83 ± 0.17 | 0.61 ± 0.05 | 19.53 ± 1.12 |

**Table 4. Standard errors for average abundance per survey of foraging guilds before (1997) and after (2002) restoration at Harrier Meadow.**

| Foraging Guild | 1997 | 2002 |
|---|---|---|
| Aerial | absent | 1.08 |
| Generalist | 2.29 | 18.69 |
| Mudflat | 0.52 | 188.91 |
| Open water | 1.22 | 12.39 |
| *Phragmites* | 1.26 | 0.76 |
| Upland | 0.89 | 1.35 |

**Table 5. Species richness of six foraging guilds at Harrier Meadow before and after restoration, including number of species gained and lost, and Sorensen's similarity index for pre- and post-restoration species composition.**

| | Upland | *Phragmites* | Open water | Mudflat | Generalists | Aerial | **Total** |
|---|---|---|---|---|---|---|---|
| Pre-restoration | 17 | 3 | 13 | 1 | 8 | 0 | 42 |
| Post-restoration | 13 | 3 | 22 | 6 | 11 | 2 | 57 |
| Gained post | 3 | 0 | 11 | 5 | 5 | 2 | 26 |
| Lost post | 7 | 0 | 2 | 0 | 2 | 0 | 11 |
| Similarity | 0.67 | 1.0 | 0.63 | 0.29 | 0.63 | 0.0 | |

**Table 6. (a) Avian species present before and after restoration at Harrier Meadow, with a frequency of = 0.67. (b) Species present in all three seasons before and after restoration at Harrier Meadow.**

**a.**

| Species with frequency = 0.67 | |
|---|---|
| Pre-restoration | Post-restoration |
| Red-winged blackbird | Snowy egret |
| Song sparrow | Mallard |
| Swamp sparrow | Killdeer |
| | Song sparrow |
| | Great egret |
| | European starling |
| | Canada goose |
| | Red-winged blackbird |
| | Marsh wren |
| | Great black-backed gull |
| | Mute swan |
| | Ring-billed gull |

**b.**

| Species present in all three seasons | |
|---|---|
| Pre-restoration | Post-restoration |
| American goldfinch | American goldfinch |
| European starling | Barn swallow |
| Great egret | Black-crowned night heron |
| Marsh wren | Canada goose |
| Ring-necked pheasant | *Calidris* species |
| Red-winged blackbird | Double-crested cormorant |
| Snowy egret | European starling |
| Song sparrow | Gadwall |
| Swamp sparrow | Great black-backed gull |
| | Great blue heron |
| | Great egret |
| | Herring gull |
| | Killdeer |
| | Mallard |
| | Marsh wren |
| | Mourning dove |
| | Mute swan |
| | Northern mockingbird |
| | Ring-billed gull |
| | Rock pigeon |
| | Red-winged blackbird |
| | Semipalmated plover |
| | Snowy egret |
| | Song sparrow |
| | Spotted sandpiper |
| | Swamp sparrow |
| | *Tringa* species |

**Table 7. Average (± SE) avian diversity, evenness, and species richness per survey during the fall and spring migration and summer breeding seasons at Harrier Meadow.**

| | **Diversity** | | **Evenness** | | **Species Richness** | |
|---|---|---|---|---|---|---|
| | **1997** | **2002** | **1997** | **2002** | **1997** | **2002** |
| **Spring** | 1.79 ± 0.13 | 2.12 ± 0.18 | 0.74 ± 0.03 | 0.68 ± 0.05 | 11.6 ± 1.60 | 22.6 ± 1.03 |
| **Summer** | 1.83 ± 0.12 | 1.85 ± 0.30 | 0.80 ± 0.04 | 0.61 ± 0.10 | 9.8 ± 0.80 | 21.4 ± 1.60 |
| **Fall** | 1.78 ± 0.19 | 1.51 ± 0.35 | 0.86 ± 0.03 | 0.56 ± 0.13 | 8.6 ± 1.63 | 14.6 ± 0.51 |

**Table 8. Average (± SE) avian diversity. evenness, and number of species per hectare at Harrier Meadow and Mill Creek after restoration.**

| | Diversity | | Evenness | | Species richness | |
|---|---|---|---|---|---|---|
| | Harrier | Mill Creek | Harrier | Mill Creek | Harrier | Mill Creek |
| 2001 | 1.71 ± 0.16 | 1.76 ± 0.15 | 0.56 ± 0.05 | 0.65 ± 0.05 | 2.28 ± 0.14 | 1.07 ± 0.06 |
| 2002 | 1.83 ± 0.17 | 1.74 ± 0.16 | 0.61 ± 0.05 | 0.66 ± 0.06 | 1.99 ± 0.11 | 1.01 ± 0.04 |
| 2003 | 1.91 ± 0.15 | 1.67 ± 0.14 | 0.65 ± 0.05 | 0.65 ± 0.05 | 2.01 ± 0.16 | 0.91 ± 0.04 |
| Average | 1.81 ± 0.09 | 1.72 ± 0.08 | 0.61 ± 0.03 | 0.65 ± 0.03 | 2.10 ± 0.08 | 1.00 ± 0.03 |

# History of the Eastern Screech-Owl (*Megascops asio*) in New York City, 1867–2005*

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## Abstract

This paper provides a historical review of the occurrence, distribution, and changes in abundance of the eastern screech-owl (*Megascops asio*) in New York City, New York. From the late 19th century through the mid-20th century, this owl species was a common permanent member of the avifauna throughout New York City. By the 1960s, information about eastern screech-owls was much less common in New York City bird reports, suggesting the species had declined in the area. In 2005, this owl was believed to be breeding in three of five boroughs of the city but was common only on Staten Island. I discuss the eastern screech-owl restoration efforts in Central Park, Manhattan. Eastern screech-owls nested and fledged young in March 2002 and March 2005—the first confirmed nesting of this species in Central Park since 1949. However, the probability of their long-term survival in Central Park is uncertain. Recommendations are made to facilitate future releases of this species in New York City.

**Keywords**: Eastern screech-owl; *Megascops asio*; New York City, Central Park; restoration; urban

## Introduction

The eastern screech-owl (*Megascops asio*; Figure 1) is the most common owl in eastern North America, often nesting in tree cavities and nest boxes in close proximity to people (Gehlbach, 1995). In New York City, this small owl was formerly quite common, found nesting in all five boroughs in the first half of the 20th century. However, between 2001 and 2005 the species was documented to nest annually in only one of the city's five boroughs (Staten Island), and to probably nest annually in two others (northern Manhattan and parts of the Bronx). In order to reestablish a population in a former breeding area, the New York City Department of Parks and Recreation released rehabilitated eastern screech-owls into Central Park, in Manhattan, in 1998 and 2001–02. This paper summarizes what is known of the eastern screech-owl in New York City since the latter half of the 19th century and discusses the Central Park restoration effort.

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## Methodology

### A. Historical Distribution and Abundance in New York City

Information regarding the occurrence, number, and natural history of eastern screech-owls in New York City was gathered from 19th- and 20th-century notes and articles in journals and books; reports to New York City organizations; and recent e-mail communications with knowledgeable local bird-watchers. Data about eastern screech-owls are provided for each of the city's five boroughs. To obtain additional information about the relative abundance of eastern screech-owls in each borough in the past, Christmas Bird Count (CBC) data were compiled from 1900 through 1999. In addition, to determine the relative historical proportion of red- versus gray-morph eastern screech-owls in New York City, I examined the collection of bird specimens held at the American Museum of Natural History (AMNH).

### B. Restoration Into Central Park, New York City

I designed the project to restore the eastern screech-owl to Central Park in 1997, and I handled the 1998 release. The 2001–2002 introduction was conducted by William Giuliano and Christopher Nagy, then at Fordham University (Nagy, 2004). A total of 38 eastern screech-owls were released in Central Park over the course of four years (1998: n = 6; 2001: n = 18; 2002: n = 14). All the owls were obtained from raptor rehabilitators and were = one year old when released. In August 1998, six eastern screech-owls obtained from the Raptor Trust in New Jersey were released. An additional 18 owls were released in September through October 2001, 2 in March 2002, and 12 in August 2002. These came from raptor rehabilitators in central Michigan and upstate New York. The owls obtained from Michigan were color-marked on their faces. A "hard" release method was used: Owls were brought to the park late in the day of release, banded, and set free at dusk. Once set free, no food was provided to the owls at feeding stations.

Central Park (40° 47' N, 73° 58' W), the habitat into which the owls were released, is a 344-hectare (860-acre) public space in the center of Manhattan. There are two primary woodlands in the park: a 15-hectare (38-acre) parcel called "the Ramble" in the center of the park, and a 36-hectare (90-acre) tract in the north end referred to as "the North Woods" (Fowle & Kerlinger, 2001). These two woodlands are composed of deciduous tree species including native black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), and red oak (*Quercus rubra*), as well as nonnative black locust (*Robinia pseudoacacia*) and Norway maple (*Acer platanoides*). Both woodlands are heavily transected by pedestrian paths. The ground layer has been greatly affected by different types of disturbance, with the result that nonnative herbaceous species dominate large areas of the forest floor (Loeb, 1993). In order to increase the number of roosting/nesting cavities, 15 cedar-wood owl boxes were placed in these woodlands in June 1998.

The author did not carry out a small-mammal survey or any other type of prey-base analysis as part of the original restoration project. It was inferred that the prey base in Central Park was sufficient because each winter from 1990 to 1998, up to five long-eared owls (*Asio otus*), as well as saw-whet owls (*Aegolius acadicus*) and occasionally other owl species, were present concurrently in the park. Each of these raptor species preys upon the same small mammals that eastern screech-owls prey upon. Eastern screech-owls also capture a variety of other prey, such as small birds like the house sparrow (*Passer domesticus*; see

Nichols, 1953) and invertebrates (Sutton, 1929), and these are common in Central Park for most of the year.

In order to determine prey items consumed by eastern screech-owls in Central Park, I collected owl pellets whenever possible from October 2001 through October 2002. A total of 51 pellets were collected and analyzed in 2001–02. Thirty-nine pellets were recovered in the Ramble and environs, and 12 were collected from the North Woods. Twenty-nine pellets collected from August through late October 2002 came from the same pair of red-morph owls that roosted near the Ramble. The 12 pellets collected from the North Woods area came from one gray-morph individual.

To determine how many individuals should initially be released in Central Park in the restoration project, I researched published home-range sizes in suburban areas where moderate to high levels of food were also available to eastern screech-owls (see Smith & Gilbert, 1984; Gelhbach, 1994). Owl home ranges varied from between 11 to 131 hectares in a suburban Connecticut study (Smith & Gilbert), so I estimated that Central Park could support up to five pairs of breeding eastern screech-owls. Also, research from other studies has shown that normal mortality of approximately 60% to 75% could be expected for eastern screech-owls = 1 year old (VanCamp & Henny, 1975). It seemed reasonable to release six owls in 1998 as a test to determine if rehabilitated owls could survive in an urban habitat. In 2001–02, it was believed a greater number (up to 25) of eastern screech-owls should be released simultaneously each year in order to establish breeding pairs as quickly as possible in Central Park (DeCandido, personal observation, 2002; Nagy, 2004).

After the releases, some information about the location, behavior, and territories of the Central Park eastern screech-owls came from the author's field notes made during diurnal searches on foot, as well as through playback of the territorial ("whinny") call at night. These data records were made throughout the year from 12 August 1998 through 1 March 2003, and again from January 2004 through December 2005. Giuliano and Nagy monitored the owls and collected data from mid-September 2001 to August 2003 using radiotelemetry and other methods (Nagy, 2004).

## Results

### A. Historical Distribution/Abundance in New York City, 1867–2005

In the mid- to late 19th century, the eastern screech-owl was known to breed in two boroughs of New York City: the Bronx and Manhattan. Eugene P. Bicknell (see Griscom, 1926) considered this species to be a common permanent resident in the Riverdale area of the Bronx at that time. In Manhattan, as part of a report to the Board of Commissioners of Central Park (Anonymous, 1869), the eastern screech-owl was described as "permanent resident; abundant; build their nest[s] in the crevices of the rocks in the Ramble." However, almost 20 years later, Woodruff and Paine (1886) listed the owl as "resident; not common" in Central Park. In Brooklyn, Wyman (1883) does not record the presence of this species in Prospect Park, though one eastern screech-owl was collected in 1867 in that borough (AMNH # 437303). No 19th-century data regarding the occurrence of this species could be obtained for Queens and Staten Island.

In the Bronx in the early 1900s, eastern screech-owls were reported from Riverdale and Van

Cortlandt Park by E.P. Bicknell (see Griscom, 1926). Kuerzi (1926) listed the owl as a "common permanent resident" of the Bronx. In a winter survey in 1953–54, Buckley (1958) found seven owl species in the Bronx (Pelham Bay Park), but he did not find the eastern screech-owl. In a single-night survey for owls on 8 December 1956 (Buckley, Carleton, Post & Scully, 1960), eastern screech-owls were found along the Bronx River on the grounds of the New York Botanical Garden. From 1957 to 1987, this owl was found on 10 of 31 Christmas Bird Counts in Pelham Bay Park (I. Cantor, unpublished data, 2000). Between 1998 and 2005, eastern screech-owls were reported from Riverdale Park (C. Jaslowitz, personal communication, 2004), Van Cortlandt Park (D. Kunstler, personal communication, 2004), and along the Bronx River (E. Edler, personal communication, 2005). The owl is presumed to still be breeding in these three areas of the Bronx, but it is much less common than in the recent past.

In Central Park in 1908, the eastern screech-owl was one of 18 breeding bird species found that year (Griscom, 1925). By 1924, when there were only eight native breeding bird species known in Central Park, there were still "several resident pairs" of eastern screech-owls (Griscom, 1925). In the late 1940s, Carleton (1947), summarizing the status of all birds in Central Park, wrote that the eastern screech-owl was "seen almost every year. No summer records in recent years." For the period between 1948 and 1957, Carleton (1958) wrote that the owl was a "permanent resident, occasionally found breeding." By 1970, Carleton no longer listed this species as part of the avifauna of the park, except for a single eastern screech-owl seen on the 1955 Christmas Bird Count (Carleton, 1970). The last literature citation of extant eastern screech-owls in Central Park was Bull (1964), who stated that the species still nested in the early 1960s, but no specific information was provided. Knowler (1984), in a yearlong survey of Central Park in 1982, does not mention this species as having been seen. Kerlinger and Sanford (1998) could not locate any evidence of eastern screech-owls (or the presence of any owl species) in a comprehensive breeding-bird survey of Central Park conducted from 23 May through 17 July 1998. In northern Manhattan, at Inwood Hill Park, no written historical records could be found that described the status of this species in either the 19th or 20th centuries. However between 1997 and 2005, it was still possible at Inwood Hill Park to attract eastern screech-owls using recordings, with a maximum of six (two adults and four fledglings) seen simultaneously in the first week of September 1997 (M. Feller, personal communication, 2002; see also Hellman, 1998).

In Brooklyn (Kings County), Vietor and Vietor (1909) did not record the eastern screech-owl as a breeding species in their one-year bird survey of Prospect Park, but they do mention that one individual was seen on 20 December, 1908. Walsh (1926) considered the species to be a permanent (year-round) resident in the park and wrote that a few pairs were definitely known to have nested within the borders of Prospect Park from 1908 to 1925, inclusive. Carleton (1958) listed the eastern screech-owl in Prospect Park as "permanent resident, breeds." However, by 1970 he did not list the species as occurring in the park (Carleton, 1970). The last mention of extant eastern screech-owls in Brooklyn was Bull (1964), who stated that the species still nested in Prospect Park in the early 1960s, but provided no other details.

In Queens, pre-1900 information could not be found. For the period 1915 to 1950, the most detailed information comes from the naturalist Sam Yeaton (1992):

> The common nester in our streets and backyards was the [eastern] screech-owl. In 1919, people in Flushing were familiar with owls (barn [*Tyto alba*], long-eared, and screech), and no one disturbed the screech-owls. For example, there was one in a hole in a maple about twelve feet above the ground on the corner of Sanford Avenue and Kissena Boulevard in front of St. Joseph's Home, and a sign nailed to the tree called it to the attention of all passersby and said, "Please do not disturb this owl." Squirrels on the other hand were rare. A friend of mine, seeing a squirrel in his neighborhood, made a house and nailed it to a tree in his backyard. Immediately, he got a screech-owl that lived there for many years. A screech-owl also lays four eggs in a well-protected hollow tree and usually fledges all four, much less subject to predation than baby Robins [*Turdus migratorius*].... However in 1919, screech-owls, while perhaps not abundant, were actually plentiful. And this was for many years. I remember one Christmas Count after World War II when Frank and Norton Smithe counted 13 screech-owls in Douglaston alone. There were many more red-morph than gray-morph owls, but both were present. I have a photo I took in 1924 of Harrison Skeuse holding a gray-morph screech-owl, but both were present. These were taken out of two holes in two adjacent apple trees at the south end of the gully at Oakland Lake.

Between 1999 and 2002, one red-morph individual had occasionally been seen in Alley Pond Park (H. Roth, personal communication, 2003), and a pair was seen together in spring 2005 (A. Ott, personal communication, 2005). In Forest Park, a red-morph owl was present in summer to fall 2002, and a gray-morph individual was seen in May 2004 (E. Lam, personal communication, 2004; A. Ott, personal communication, 2004). No eastern screech-owls have ever been seen at the Jamaica Bay Wildlife Refuge since its inception in 1953 through August 2005 (D. Riepe, personal communication, 2005).

On Staten Island (Richmond County), the eastern screech-owl was known to naturalists in the 19th century, but no further data were recorded (Davis, 1892; Siebenheller, 1981). A.C. Bent (1938) wrote that an individual was banded on Staten Island on 27 October 1925 and recaptured in the same place some eight years later, on 11 July 1933. In the 1970s, nests were found in High Rock Park, Lighthouse Hill, and in Sunnyside, but these were only a few pairs of the total breeding population at that time (Siebenheller, 1981). The maximum number of nests reported in one area of Staten Island was five at Blue Heron Park, in 1990 (D. Riepe, personal communication, 2003). Between 2001and 2005, eastern screech-owls were still known to nest in several areas such as Wolfe's Pond Park, Blue Heron Park, the Greenbelt, Long Pond Park, William T. Davis Wildlife Refuge, Conference House Park, and the town of Princess Bay (R. Matarazzo, personal communication, 2004; S. I. Wollney, personal communication, 2004). They may still nest on Grymes Hill/Sunnyside, the Silver Mount Cemetery area, the Moravian Cemetery, and scattered locales along the west and south shores of Staten Island (H. Smith, personal communication, 2004; M. Shanley, personal communication, 2004). However, since the 1990s, eastern screech-owl populations on Emerson Hill and the St John's/Notre Dame area have probably been extirpated due to development (M. Shanley, personal communication, 2004). The future of this species is more secure on Staten Island than in any other borough.

### B. New York City Christmas Bird Count Records, 1900–1999, and American Museum Specimens, 1867–2005

According to Table 1, Christmas Bird Count (CBC) records for the period 1900–1999 show that eastern screech-owls have been seen in every borough in New York City. The maximum number recorded was 13 on the Bronx-Westchester CBC in 1956. Other single-year-high counts were recorded in Queens (10 in 1950) and Staten Island (11 in 1971). Nonreleased eastern screech-owls have not been found on area CBCs in many years—not since 1955 in Manhattan, 1962 in Brooklyn, and 1965 in Queens.

The majority (83.3%) of 18 eastern screech-owls collected in New York City and held at AMNH have been red-morph individuals (Figure 1). Most of the individuals collected in New York City came from Brooklyn (46.7%), followed by Queens (26.7%) and Staten Island (20%). No AMNH specimens have ever been collected in Manhattan or the Bronx. The earliest AMNH-collected eastern screech-owl came from Brooklyn (1867), and most specimens (61.1%) were collected from 1889 to 1915.

### C. Observations of Released Eastern Screech-Owls in Central Park, 1998–2005

Of the 1998 group, one owl was injured in a collision with a car approximately 30 days after release and was returned to the Raptor Trust. On 23 December 1999, an owl was found injured and died soon after. Toxicology analysis showed no detectable (or significant) levels of rodenticides, herbicides, or insecticides (Stone, 2000). Only one eastern screech-owl could be found in Central Park during intensive nocturnal surveys in July 2001 by the author. The fate of three others released in August 1998 remains unknown. By August 2002, no evidence could be found in Central Park of any of the original six owls released in 1998. Of the owls released in the 2001–2002 study, seven individuals were thought to be alive as of 2004 (Nagy, 2004). During diurnal and nocturnal surveys of Central Park in November–December 2005, I found at least three pairs of eastern screech-owls occupying different areas of Central Park: one pair each on the east and west side of the Ramble, and one pair in the North Woods. The North Woods pair consisted of two gray-morph birds. The pair (both gray-morph birds) on the west side of the Ramble was observed leaving a shared roost cavity on several occasions beginning in late November 2005. The pair (gray-morph male and red morph female) on the east side was first observed leaving a shared roost cavity in early December 2005. In addition to these three pairs, it is estimated that there were up to five other eastern screech-owls residing in Central Park in November–December 2005.

On 28 March 2002, two fledgling eastern screech-owls were discovered near their nest site near the Ramble (Forbes, 2002). The parents of these young were a female released in 1998 and a male released in September 2001. Neither parent had a backpack transmitter when observed in 2002. On 19 March 2005, three fledglings were discovered by birders in the northern part of Central Park (Figure 2). Both these fledging dates are more than 30 days earlier than fledging dates previously recorded for this owl species in New York State (Andrle & Carroll, 1988). A historical record from Queens County indicates that eastern screech-owls fledged young on at least one occasion in June (see Anonymous, 1934).

None of the owls released in 1998 were ever found using a cedar nest box for roosting. In Central Park, preferred roost-tree cavities were in native black cherry, hackberry, and (rarely) red oak, as well as nonnative black locust and London plane tree (*Platanus* × *acerifolia*). In mild weather, eastern screech-owls could often be found roosting on branches of Norway maple, where they were concealed by the foliage, and occasionally on conifers such as eastern hemlock (*Tsuga canadensis*) and white pine (*Pinus strobus*). Most roost sites were between 3 and 6 meters from the ground, and the highest was approximately 21 meters. The lowest diurnal roost site was below a partially fallen tree near a rock outcropping, approximately 15 centimeters off the ground.

Analysis of 10 eastern screech-owl pellets collected from the Ramble area of Central Park from late August 2001 through February 2002 showed that six small birds, four small rodents, one fish, and an arthropod (possibly a crayfish) had been eaten. In late February 2002, analysis of 12 pellets recovered from one individual in the North Woods contained 15 small rodents and 2 birds. From August through October 2002, analysis of 29 pellets recovered near the Ramble showed that at least 13 birds and 22 small mammals had been consumed. Numerous arthropod parts were also recovered from each of these pellets, and invertebrates were especially common in pellets collected from 3 August through 10 September 2002. In the necropsy of the eastern screech-owl found on 23 December 1999, Stone (2000) found that it had a stomach full of 14 green lepidopteran caterpillars. In the necropsy of an adult female eastern screech-owl found dead in late January 2002 from poisoning with brodifacoum, an anticoagulent rodenticide, Stone (2002) wrote that the owl was in good flesh and had abundant fat. In June 2002, a pair of red-morph owls were observed capturing flying insects such as fireflies (Coleoptera: Lampyridae) on a lawn on the east side of the Ramble at dusk (see also Sutton, 1929). In March–April 2005, the three fledgling and parent eastern screech-owls were observed feeding upon large, nonnative (Asian) earthworms on lawns in the northwestern section of the park on several occasions (J. Demes, personal communication, 2005).

## Discussion

New York City has lost two nocturnal and three diurnal raptors as nesting species in the last century. Both the long-eared owl and the barred owl (*Strix varia*) have been extirpated as breeders since 1900 (Table 2). The eastern screech-owl has also declined throughout much of its former range in New York City, even in areas protected as parkland. Historical information combined with data from CBCs indicates that eastern screech-owls were breeding in every borough in New York City from at least the mid-19th century through about 1955. In the mid- to late 1950s, the eastern screech-owl was extirpated from Central Park, and in the early 1960s, from Brooklyn (Prospect Park). Observations made and reports received from bird-watchers from 2001 to 2005, combined with recent CBC data, suggest that the eastern screech-owl was still breeding annually on Staten Island, and probably breeding annually in Inwood Hill Park in northern Manhattan, and in several parks in the Bronx.

It is difficult to determine why the eastern screech-owl was extirpated from several parks in New York City since 1950. Small, isolated populations such as those in Central and Prospect parks were vulnerable to extirpation due to a variety of causes (often acting in concert), including stochastic (chance) events and reduced gene flow. Local extirpation factors may have included (a) the increased use of anticoagulant rodenticides (primarily brodifacoum and bromadiolone) and insecticides

(DDT) beginning in the 1950s; (b) removal of dead trees and snags from city parks; (c) changes in habitat (via succession of meadows to shrubs/forest) and/or conversion of meadows/forest edges to low-cut grass lawns affecting small-mammal populations; (d) collisions with fast-moving vehicles beginning in the 1950s; (e) an increase in competitors for tree cavities such as eastern gray squirrels (*Sciurus carolinensis*), raccoons (*Procyon lotor*), and European starlings (*Sturnis vulgaris*); (f) an increase in predators such as American crows (*Corvus brachyrhynchos*); (g) predation/disturbance by nocturnal mammals such as raccoons at owl nest sites during the breeding season; (h) and the increased use of city parks by people causing undue disturbance near nest sites. Both the eastern screech-owl and the American kestrel (*Falco sparverius*) were extirpated as breeding species from Central and Prospect parks in the late 1950s or early 1960s (Bull, 1964; Carleton, 1958), but the ecologically similar, cavity-nesting American kestrel was still found breeding within several blocks of these two parks between 1995 and 2005. This suggests that a lack of high-quality nest cavities may be the most important factor limiting the nesting success of eastern screech-owls in Central Park. We believe that competition for, and disturbance at, tree cavities from high numbers of eastern gray squirrels, combined with predation upon roosting owls by raccoons, were important factors that prevented more owls from nesting successfully in Central Park from 1998 to 2005.

The two successfully nesting pairs of eastern screech-owls that fledged young in March of 2002 and 2005 in Central Park did so earlier in the year than any other pair known to nest in New York State (Andrle & Carroll, 1988.) According to nesting-cycle details published in Bent (1938) and Gehlbach (1995), these two Central Park pairs laid eggs from 20 January –30 January. Also, Stone (2002), in a necropsy of a female owl found dead in Central Park on 22 January 2002, wrote that "the oviduct was enlarged and thickened for egg laying and a brood patch seemed to be developing." In eastern North America, eastern screech-owls lay eggs primarily in March and April (Bent, 1938; Gehlbach 1995). In Central Park, these early nest records suggest that there are factors such as the urban heat island effect, abundant artificial light, and high levels of food that greatly influence eastern screech-owl reproductive cycles.

Few studies have been done to determine what happens to released rehabilitated raptors, especially in an intensive program of restoration of first-year birds such as the ones conducted in Central Park. (For information on such releases, see Bennett & Routh, 2000; Csermely, 2000.) Long-term results from the Central Park study will provide important information to raptor rehabilitators, who frequently do not know what happens to their birds once they are released back into the wild. In order to keep track of the released eastern screech-owls in Central Park, a combination of three types of survey techniques was most effective in locating owls: walking the park on a regular basis during the day; walking the park at night (while playing recordings of owl calls) at least once per week; and radiotelemetry tracking. Color-marking the face of released owls proved invaluable in determining the exact identity of individuals that perched at the entrance of tree cavities (see Figure 3).

One of the great advantages of doing species restorations in urban parks is the opportunity it affords to work with the local community as well as environmental groups such as the Nature Conservancy, the Brooklyn Bird Club, the Wildlife

Conservation Society, and New York City Audubon. A cadre of citizen-scientists can monitor owls on a regular basis. More important, regularly scheduled owl walks put the community of longtime bird-watchers in touch with new (often younger) ones. It is then possible to convey good owling etiquette to newcomers.

That eastern screech-owls have been extirpated from areas of their former metropolitan range is part of a larger issue: the loss of native plants and animals in New York City parks. The most important lesson to be learned in this restoration project is that it is much easier to preserve and protect species already living in parks than it is to reestablish species once they have been eliminated. From a broader perspective, much of New York City's natural heritage is being lost at an alarming rate (DeCandido, Muir & Gargiullo, 2004). Though scientists tend to interpret species extinctions in terms of biological processes, the future of the native fauna and flora of New York City depends on viewing the issue from a different perspective. The critical factor in preserving species diversity is developing public support for natural areas in parks. Much more effort needs to be devoted to explaining why preserving native species is important (Tilman, 2000). The degree to which scientists create opportunities for people to appropriately enjoy the remaining natural areas in urban parks will help determine the future of native species within them, and the natural areas themselves. That several environmental groups and numerous individuals have come together via the eastern screech-owl restoration project points to some measure of hope in this endeavor.

## Management Recommendations

Programs to reintroduce eastern screech-owls in New York City should strive to take full advantage of available media outlets, with one important caveat: The restoration must adhere to a plan designed and supervised by Ph.D. biologists in order to maintain the scientific integrity of the endeavor. I recommend that scientists from the Wildlife Conservation Society supervise any future releases in the city.

As part of the restoration plan, the public should be involved in release as well as post-release activities—especially in monitoring the owls. Allowing people in the birding community to participate in the actual release of the birds makes them an integral part of the process and reinforces the notion that they are stewards of the owls (DeCandido & Allen, 2002).

In New York City, two other areas are appropriate for an eastern screech-owl restoration/introduction: Prospect Park in Brooklyn and the Jamaica Bay Wildlife Reserve (JBWR) in Queens. Efforts to establish eastern screech-owl populations should focus on these parks, especially JBWR, where the habitat, food base, and interest from bird-watchers and media are optimal. Important landscape-management issues should be discussed with park managers before release, including the use of broudifacoum as a rodenticide, the pruning of dead limbs and trees, and the provision of numerous cedar nest boxes. Consideration should be given to providing food for the owls at feeding station(s) for several weeks after the owls are released.

If it becomes necessary to release additional eastern screech-owls to augment the population in the park, one method would be to partner with the Central Park Zoo (Wildlife Conservation Society). Since 2001 the zoo has had a captive pair of eastern screech-owls on display, along with a descriptive text of the owl-restoration project in the park. If its two captive owls could breed, and if the young were to fledge directly into Central Park, it would be an easy way to increase the number of young owls each year and to increase genetic diversity of the existing population in the park.

Finally, New York City is the media capital of the world, and this should be used to the advantage of the project and participating organizations. Biologists must be ready with a specific conservation message to impart to the general public. Careful thought should be given as to the structure of the message, as well as to the information it provides. Currently, there is no printed information available for the general public describing the Central Park eastern screech-owls or the restoration project. I recommend that an educational brochure containing detailed information about the history of the eastern screech-owl in New York City be developed as quickly as possible.

## Acknowledgments

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## Glossary

**Arthropod:** An invertebrate animal belonging to the phylum Arthropoda, such as an insect or crustacean.

**Avifauna:** The birds of a particular region or time period.

**Gene flow:** The spread of genes through populations as affected by movements of individuals and their propagules (e.g., plant seeds). (*Penguin Dictionary of Biology*)

**Necropsy:** An examination and dissection of a dead body to determine cause of death or the changes produced by disease. (*Wordnet*; Princeton University)

**Figure 1. Eastern screech-owl, red-morph. Photo courtesy of the Raptor Trust, New Jersey.**

**Figure 2. Fledgling eastern screech-owls, Central Park, New York City, 26 March 2005. Photo © 2005 Deborah Allen.**

**Figure 3. Eastern screech-owl (*Megascops asio*), gray-morph, in Central Park, New York City. Photo © 2001 Deborah Allen.**

**Table 1. Eastern screech-owls (*Megascops asio*) recorded on 100 years of Christmas Bird Counts (CBC) by borough in New York City from 1900 to 1999.**

| Borough (year CBC began) | **Year First Found (number)** | **Year Max (number)** | **Year Last Found (number)** | **Number of Years Found (%)** |
|---|---|---|---|---|
| Bronx (1902) | 1924 (1) | 1956 (13) | 1999 (3) | 68/98 = 69.4% |
| Brooklyn (1904) | 1908 (1) | 1960 (3) | 1962 (1) | 18/96 = 18.8% |
| Manhattan (1900) | 1931 (1) | 1948 (1) | 1955 (1) | 3/100 = 3.0% |
| Queens (1903) | 1933 (6) | 1950 (10) | 1965 (1) | 20/97 = 20.6% |
| Staten Island (1910) | 1914 (1) | 1971 (11) | 1999 (5) | 47/90 = 52.2% |

**Table 2. Status of nesting and wintering owls throughout New York City from 1995 to 2005**

| Species | Nesting | Winter Resident |
|---|---|---|
| Barn Owl* (*Tyto alba*) | Common | Common |
| Eastern Screech-owl* (*Megascops asio*) | Uncommon | Uncommon |
| Great Horned Owl* (*Bubo virginianus*) | Common | Common |
| Snowy Owl (*Bubo scandiacus*) | --- | Rare/Uncommon |
| Barred Owl (*Strix varia*) | Extirpated | Rare |
| Long-eared Owl (*Asio otus)* | Extirpated | Common |
| Short-eared Owl (*Asio flammeus*) | --- | Rare/Uncommon |
| Northern Hawk Owl (*Surnia ulula*) | --- | Extremely Rare[1] |
| Northern Saw-whet Owl (*Aegolius acadicus*) | --- | Common |
| Boreal Owl (*Aegolius funereus*) | --- | Extremely Rare[2] |

**Key to Status: Nesting**

**Common:** More than 10 nests known in New York City in a given year; or, nests in several places in two or more boroughs, 1995–2005.

**Uncommon:** Fewer than 10 total nests known in New York City in a given year; or, a common nesting species in only one borough (e.g., the eastern screech-owl in Staten Island), 1995–2005.

**Extirpated**: Formerly bred in New York City. The barred owl was last found nesting in New York City in Staten Island in 1908.The long-eared owl was last found nesting in Staten Island in 1947.

**Key to Status: Winter**

**Common:** Found from December through February in appropriate habitat at night or known diurnal roosts every year in one and usually more boroughs, 1995–2005.

**Uncommon:** Not likely to have been found by experienced birders from December through February in 1995–2005 in the appropriate habitat at night or at known diurnal roosts; or, common only in one borough.

**Rare:** Found fewer than five times per season by experienced birders in appropriate habitat in New York City from 1995 to 2005.

**Extremely Rare:** Single Records: (1) northern hawk owl—collected in 1863 (no date) in Brooklyn, AMNH Collection # 437332; (2) boreal owl—observed Central Park, Manhattan, from 19 December 2004 until 14 January 2005.

* Nested in New York City in 2001–05.

# Bringing the Urban Environment Into the Classroom: Learning From an Estuarine Mesocosm*

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## Abstract

As the United States and the world become increasingly urbanized, human populations exert a more concentrated impact on their local environments. The effects of human activity extend well beyond the geographical borders of cities, to almost every remote area. This worldwide urbanization has the additional impact of distancing urban youth from pristine habitats and making it difficult for them to connect with the natural world. This paper describes an inquiry-based educational unit that is designed for an environmental science class, biology class, or general science class and can be taught at a variety of grade levels (grades 6–12) with slight modification. In the unit, which supports National Science Education Standards (Appendix A), small groups of students observe an estuarine mesocosm. Each mesocosm is seeded with one common macroinvertebrate that inhabits urban salt marshes. Students research their organism through observation and literature review and present their findings to the rest of the class. The purpose of the unit is manifold: 1) to allow students to reconnect with the natural world; 2) to introduce the concept of adaptation to the urban biome; 3) to allow students to understand that many commonly encountered organisms may have economic or recreational benefits to human society as well as value independent of human concerns; and 4) to increase students' knowledge base regarding salt marsh ecosystems and the natural histories of four salt marsh–inhabiting organisms. Requiring the students to conduct independent research and report their findings to the class engages them in peer teaching and also forms a basis for formative and summative assessment. In addition, because the unit may require the use of multiple computer programs by the students, it reinforces or introduces the use of such tools in a format that is likely to maintain their interest.

**Keywords:** urban ecology, lesson plan, secondary school, education, starlet sea anemone, banded killifish, grass shrimp, periwinkle

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## Introduction

As of April 1, 2000, 79% of Americans lived in urban environments (U.S. Census Bureau, 2002). Worldwide, urban populations are growing rapidly, in both absolute and relative terms. The percentage of the world's population living in urban areas is expected to rise to 58% by 2020 (McDevit, 1996). The recent increase in density and size of urban areas is attributed to increased fertility, decreased mortality, and population redistribution. As the United States and the world become increasingly urbanized, human populations exert a more concentrated impact on their local environments. The effects of human activity extend well beyond the geographical borders of cities, to almost every remote area. This worldwide urbanization has the additional impact of distancing urban students from pristine habitats and making it difficult for them to connect with the natural world.

It is imperative that students be introduced to urban ecology as part of an environmental science or biology curriculum. Urban students are already superficially familiar with the urban biome through their own experience. An academic investigation of this familiar biome provides an opportunity to teach many fundamental concepts of biology and environmental science. In this way, this lesson in urban ecology exemplifies a "place-based education" schema. The lesson can easily be integrated as a curriculum piece in an ecology, biology, or environmental-science class. Teaching students about the natural histories and biology of urban organisms helps them to understand biological concepts that extend well beyond those organisms (for example, the challenges of maintaining homeostasis in an urban aquatic environment). Teaching students about how organisms adapt to the urban biome, such as through pollution resistance, provides a natural transition into population genetics, Hardy-Weinberg equilibrium, and evolution in a broader sense. In addition, showing the economic or recreational benefits of certain organisms allows students to understand how many environmentalists and environmental economists support conservation interests (Cropper & Oates, 1992).

This lesson unit serves as a venue for open dialogue about the philosophical value of ecosystems and the organisms they harbor. Numerous researchers have argued that biophilia (and its corollary, personal environmental responsibility) is a direct result of bonding with natural systems and organisms during childhood (Flicker, 2002; Kellert & Wilson, 1993). This connection with the natural world is difficult for urban children to forge (Pyle, 2002). In addition to its science content, this unit is designed to encourage students to reconnect with, and foster stewardship for, the natural world (Pyle, 2003).

Estuaries often provide nearby cities with protected harbors as well as inland access through rivers (Figure 1), which makes these habitats particularly susceptible to the dangers of human population. According to a 1999 population census, six of the ten largest urban areas worldwide are located on estuaries (Table 1). Their high productivity, the temporal and spatial variability in salinity, water depth, and temperature (Nixon & Oviatt, 1973; Roman, Jaworski, Short, Findlay & Warren, 2000), in conjunction with high rates of human disturbance, makes estuaries an ideal candidate for an urban ecology lesson.

In the lesson, groups of students observe mesocosms of a salt marsh environment. Each group is assigned one organism (killifish, *Fundulus heteroclitus*; starlet sea anemone, *Nematostella vectensis*; grass shrimp, *Palaemonetes pugio*, or periwinkle, *Littorina* species). The assigned organism is seeded in the group's mesocosm, and students observe the organism and conduct independent research. The students then present their findings to the rest of the class, with particular emphasis on the pressures of living in urban environments and the adaptations that might allow these species to thrive in them. Students are challenged to describe the value of each organism: its value to human society and its intrinsic value. A prepared reference sheet about each organism (Appendices C–F) may be given to the students at the discretion of those implementing this lesson. By requiring a presentation, this unit allows students to develop observational, literature-research, and presentation skills.

This inquiry-based unit follows the National Science Education Standards (Appendix A; National Research Council, 2000) and is designed to actively engage students by allowing them to pursue their own questions as well as those posed by the instructor(s).

The lesson is written for urban classes that have access to a salt marsh from which to remove sediment and sample for organisms. However, it may be easily modified to accommodate schools without easy access to a salt marsh: The animals used in the lesson can be ordered from biological supply companies and housed in tanks containing dilute artificial seawater (e.g., Instant Ocean), with or without the addition of sediment. Alternatively, the lesson may be modified for aquatic or terrestrial mesocosms seeded with organisms from local biomes.

This curriculum was designed explicitly for the school and group of students for which it was piloted and is presented with details of its implementation, but educators can modify it to meet their own particular goals, both in terms of overall structure and detail. For example, while the use of the Internet for research and Microsoft PowerPoint for presentations is suggested here to increase the interdisciplinary value of this unit, educators can substitute other research methods or assessment techniques at their discretion.

The curriculum was designed in the summer and fall of 2004 and piloted in the fall of 2004 at Odyssey High School, South Boston, Massachusetts, in two classes (Figure 2). Odyssey High School is a racially heterogeneous (Table 2) urban public high school with diverse learners.

## Prerequisite Knowledge and Skills

Prior to coming into this unit, students should acquire a basic working knowledge of their urban environment through maps, walking tours, discussion of local history, and introductory biology lessons. They should understand that every habitat has been affected in some way by anthropogenic disturbance, but that despite the human impacts, urban areas may harbor a high biomass. Students should have a rudimentary knowledge of estuarine environments in general and the role that salinity fluctuation plays in shaping the diversity in them. Familiarity with the Internet and Microsoft PowerPoint is helpful; otherwise, this lesson can be combined with an introduction to these tools.

## New Learning for Students

The content of new material the students will learn follows:

### A. Specific Content

Students will learn the i) habitat, ii) range, iii) life cycle, and iv) natural history of the following organisms, which are common inhabitants of both pristine and heavily disturbed salt marshes/estuaries:

- Banded killifish (*Fundulus heteroclitus*)
- Starlet sea anemone (*Nematostella vectensis*)
- Grass shrimp (*Palaemonetes pugio*)
- Periwinkle snail (*Littorina* species)

Each small group of students will become content expert on one species. Upon completion of observational and literature research, peer-to-peer teaching will be used to share findings and form the basis for evaluation.

**B. Skills**

Through implementation of this unit, students will learn to frame scientific questions, construct a mesocosm, perform careful observations, record data, review scientific literature, and present scientific findings.

**C. Concepts**

Students will learn that many pressures, including human disturbances such as habitat destruction and pollution, threaten the survival of urban organisms. The lesson will illustrate both common and unique strategies for survival in response to these disturbances. The effect of invasive species upon ecosystems will also be examined. Students will also learn that many organisms influence the economic or recreational value of urban environments (Cropper & Oates, 1992).

## Performance Objectives

1. Each group of students will create a mesocosm using water and sediment removed from a salt marsh pool.
2. Each group of students will perform background research and assemble a presentation using Microsoft PowerPoint software on the organism they have been assigned. This presentation will include the following content:
    - Title slide
    - Natural history information
    - Observations
    - Content specific to urban ecology:
        i. How the organism is affected by/survives in spite of human disturbance
        ii. The value of the organism intrinsically and to human society
        iii. At least two original interesting facts on or aspects of the organism "discovered" by the group during its research.

    In addition, each group will give a "tour" of its mesocosm to the other students in the class. As an alternative to the use of PowerPoint, a presentation using printed or hand-drawn transparencies or poster presentations may also be appropriate.
3. During the presentation, groups will be expected to competently answer questions posed by other students and the instructor(s).

## Materials

Materials are as follows:

- Salt marsh picture(s) (Figure 1)
- Mesocosm picture(s) (Figure 3)
- Ten-gallon aquarium per group
- Water aerators
- Five-gallon buckets
- Waders
- Shovel
- Salt marsh brackish water or Instant Ocean artificial seawater

- Project introduction sheet (Appendix B)
- Fact sheets for each organism (Appendices C–F), including a list of active websites for Internet research
- Computers with Microsoft PowerPoint software
- Projector and screen for student presentations

## Procedure

### Part I: Introduction to the Unit

This introduction requires two 50-minute classes. Time may be saved if students are not required to conduct the sampling, or if sediment is not included as part of the lesson.

The lesson begins with a highly engaging topic: mud. Literally and figuratively, sediment forms the base of the mesocosm. Mud is an engaging place to begin the discussion because most students will enter the class with the preconceived notion that mud is "gross." The students will likely be surprised to learn of its complexity and its importance in a functioning estuarine ecosystem.

1. Students are given the opportunity to view, smell, and feel a sample of highly organic sediment, though they should not be forced to smell or touch it if they don't want to (Bixler & Floyd, 1999). The instructor(s) should be aware of the quality of the sediment and ensure that all proper safety precautions are taken if the sediment is likely to be contaminated.
2. The belief that odorous sediment is "dirty" and indicates an unhealthy ecosystem should be drawn out conversationally. Questions (to ask students): "What do you think of the smell?" "Does the fact that it smells gross mean that the pool/river from which it came is unhealthy? Polluted? Dead?" (Explanations: "The highly organic nature of the sediment is responsible for the odor. The sediment is full of decomposing plants and animals," etc.)
3. After students understand the reason for the odor of the sediment, introduce the concept of a mesocosm, and provide a picture(s) of examples (Figure 3). Tie in the sediment to the picture of a mesocosm and explain that they will be developing a mesocosm of a salt marsh pool.
4. Explain the purpose and scope of the unit, providing a time line for the students if possible.
5. Brainstorm with the students to develop a list of both biotic and abiotic factors that influence the biodiversity of a salt marsh ecosystem. Make sure that students are aware that their mesocosms will contain only a small subset of the organisms that inhabit the salt marsh.
6. Formalize the list of biotic and abiotic elements of the salt marsh ecosystem. Students may be assigned homework to research and define each of these elements.
7. Brainstorm with the students to create a materials list and a procedure for sampling the sediment.
8. Formalize the materials list and procedure.

### Part II: Development of Unit

Four to six 50-minute classes will be necessary for the development of the project. Class time dedicated to the project may be reduced by requiring the students to perform work outside class hours.

At this point, the students understand what a mesocosm is; they have compiled a materials list and created a written procedure; and they know the purpose of the unit. Once any needed parental permissions have been obtained, they are ready to go to the field site.

The development of the lesson follows:

1. Students are taken to an appropriate field site. They should be divided into groups and briefed on what to expect beforehand to ensure that their educational experience in the field is not compromised by poor behavior or distractions (Crimmel, 2003). The collection site's water should have a salinity of between 5 and 15 parts per thousand (‰) to ensure survival of the species to be studied. Students are instructed to make observations about the pool from which the sediment is to be drawn and record any life forms spotted in or around the pool in their individual notebooks. They

will also measure and record the water-column depth.

An instructor enters the pool (wearing hip or chest waders) and removes sediment using a shovel. (This step will require an instructor with sufficient dexterity and strength. He/she should be aware of the proper safety precautions for the particular type of waders being utilized.) Sediment is collected in a five-gallon bucket.

Additional five-gallon buckets are filled with brackish water on site by the instructor or, preferably, by the students (safety and individuals' ability permitting).

Alternatively, if a field site visit is precluded, artificial seawater can be prepared by the instructor with a salinity of between 5‰ and 15‰.

2. Sediment and water samples are brought back to the classroom or laboratory. Each student group lines the base of an aquarium with about three inches of sediment. Water from the field site (or artificial seawater) is then added to a depth of about seven to ten inches. A small aerator can be placed in each aquarium.

3. Before distributing the organisms (collected from the field or ordered) to students, the instructor engages them in a discussion of the intrinsic value of the organisms. If the organisms are collected from the field, appropriate accommodations must be made to house the organisms at the conclusion of the lesson. (*Note*: Some *Littorina* species are not native to North American estuaries and have invasive tendencies, and *Nematostella vectensis* may also be nonnative. Do not release these animals into the wild, even if they were collected from a local estuary.)

   If they are ordered from a supplier, the instructor must be prepared to maintain the developed mesocosms. The importance of making these accommodations is discussed with the students to ensure that they realize they are stewards of their mesocosms.

   Each group is given one (or preferably more) specimen of one type of organism to study. The specimen(s) is distributed to each group in a 50 ml conical tube or some other small clear container. Ask the students to make observations about the organisms in the tubes, where they can view them in more detail than later, when the animals are in their larger environs.

   At this point, ask students questions such as "What does the organism breathe? How does the organism breathe?" These types of general questions will help focus the students on looking at systems (i.e., respiratory) analogous to their own, and will also dovetail later on with urban environmental threats (i.e., deoxygenation through anthropogenic eutrophication).

   Do not hand out the fact sheets yet. Allow students to formulate and record their own questions about the organisms. Later, they will research these questions.

4. After sufficient observation (less than 15 minutes), allow the students to place the organisms in the tanks. (Note: Before the class starts, make sure the water in the aquarium and the containers holding the organisms are roughly equal in temperature.) If using the fact sheets (Appendices C–F), hand out them out and allow the rest of the period for observations and independent reading. Depending upon the scope of inquiry the instructor wants to foster, he or she might consider not using these fact sheets at all and allowing all research and direction to be driven by the students.

5. During the next class, allow the assigned groups to organize their time, and if possible, provide students access to computers. Each student should have read the fact sheet on the organism the group has been assigned as homework or during the class before. The group needs to:

   - Conduct research (Internet, library, interviews of scientists) on the research questions/guidelines for the presentation and questions posed by the students themselves
   - Continue to make observations of the organisms
   - Develop a presentation

If the groups are not able to work independently, the time frame for each of these tasks can be scheduled. Otherwise, groups may be allowed to allocate their own resources.

**Part III: Closure of Unit**

Lectures or discussions of the pressures upon organisms in urban environments are presented, specifically addressing their adaptations to survive in an urban ecosystem, and students should be introduced to the idea of stewardship. To conclude the unit, students make a presentation on their mesocosm, which includes accepting and answering questions formulated by both other students and their instructor(s). A final homework sheet may be handed out to the students to gauge the effectiveness of the lesson and to be used in student evaluation.

## Evaluation

Throughout the project, the instructor challenges the students informally with questions to check student interest and encourage participation. Students are graded on their presentation based on written guidelines established by the individual educators and their school systems. The instructor can direct the questions during the presentation toward specific group members to aid in individual evaluation. Individual evaluation can be further determined by self-evaluations or group-peer evaluations. A homework sheet can also be used for evaluation.

## Anecdotal Results From the Lesson Pilot

While resource constraints precluded us from piloting this lesson plan in more than one school, we qualitatively found that the lesson was successful at engaging students and teaching the prescribed content. Students were surprised and intrigued by the organisms, particularly because many of them had believed that there was nothing alive in the urban estuary from which our samples were collected. There was much variation in the quality of the concluding presentations (see Figure 4 for an example of a student presentation).

## Follow-up Lessons

The presence of these organisms in the classroom/laboratory allows the opportunity to teach many other aspects of biology and environmental science. Discussions of population genetics and evolution naturally follow from discussion of adaptive resistance to toxins, which is particularly well supported in the case of *Fundulus* (Oleksiak, Churchill & Crawford, 2002; Elskus, Monosson, McElroy, Stegeman & Woltering, 1999). Interspecific competition can be illustrated between *F. heteroclitus* and *P. pugio* (Cross & Stiven, 1997). Morphology of bilaterians and radially symmetrical animals can be discussed in the context of *Nematostella vectensis* (Finnerty, 2003; Martindale, Finnerty & Henry, 2002). The same species can be used to lead an inquiry-based lesson on regeneration and/or asexual reproduction (Hand & Uhlinger, 1992; for this lesson plan see http://www.nematostella.org/Resources_Classroom_JS02.html). Other basic biology lessons utilizing this organism may be retrieved from http://www.nematostella.org/Resources_Classroom.html. Allopatric speciation can be discussed in reference to habitat fragmentation. The effects of invasive species can be illustrated by the extremely successful *Littorina* family, especially in reference to the role of humans in their dispersal (Bertness, 1984; Brenchly & Carlton, 1983).

This list of follow-up lessons is not exhaustive. Perhaps the best follow-up lessons will be those devised by students. Having aquariums in the classroom allows students to design controlled experiments to test many hypotheses. A few suggestions include predator/prey interaction experiments, growth-rate experiments, and niche-partitioning experiments.

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## Glossary

**Abiotic:** Nonliving chemical and physical parameters in an environment.

**Allopatric speciation:** The formation of new species from ancestral species as a result of geographical separation or fragmentation of a breeding population.

**Anthropogenic:** Resulting from human activity.

**Bilaterian:** Bilaterally symmetrical organism, with body parts arranged in two halves that are mirror images of one another.

**Biotic:** Pertaining to the living organisms in an environment.

**Biomass:** The total mass of living organisms, commonly of those in a particular population or ecosystem.

**Biophilia:** A word coined by Harvard biologist Edward O. Wilson to describe the deep need that people have for natural habitats and species; the love of nature.

**Estuarine:** Present in an estuary, an area where a freshwater river meets seawater and is subject to tidal fluctuations and fluctuations in salinity.

**Eutrophication:** Process by which bodies of water age and become more productive. Anthropogenic eutrophication can lead to algal blooms and deoxygenation.

**Formative assessment:** All those activities undertaken by teachers and/or their students that provide information to be used as feedback to modify and improve the teaching and learning activities in which they are engaged.

**Hardy-Weinberg equilibrium:** The law or principle that states that in an infinitely large, interbreeding population in which mating is random and there is no selection, migration, and mutation, gene frequencies will remain the same between the sexes and constant from generation to generation, with no overlap between generations.

**Homeostasis:** State in which the internal processes of an organism tend to remain balanced and stable.

**Interspecific competition:** Competition between species.

**Macroinvertebrate:** Invertebrate animal large enough to be seen without a microscope.

**Mesocosm:** A biological system used for conducting experiments.

**Niche partitioning:** The coexistence of two or more species that partition one or more resources in a habitat.

**Radially symmetrical:** With body parts arranged symmetrically around a central axis.

**Summative assessment:** The process of evaluating (and grading) the learning of students at a particular point in time.

**Figure 1. An estuary in Eureka, California**

**Figure 2. Students at Odyssey High School (South Boston, MA, 2004)**

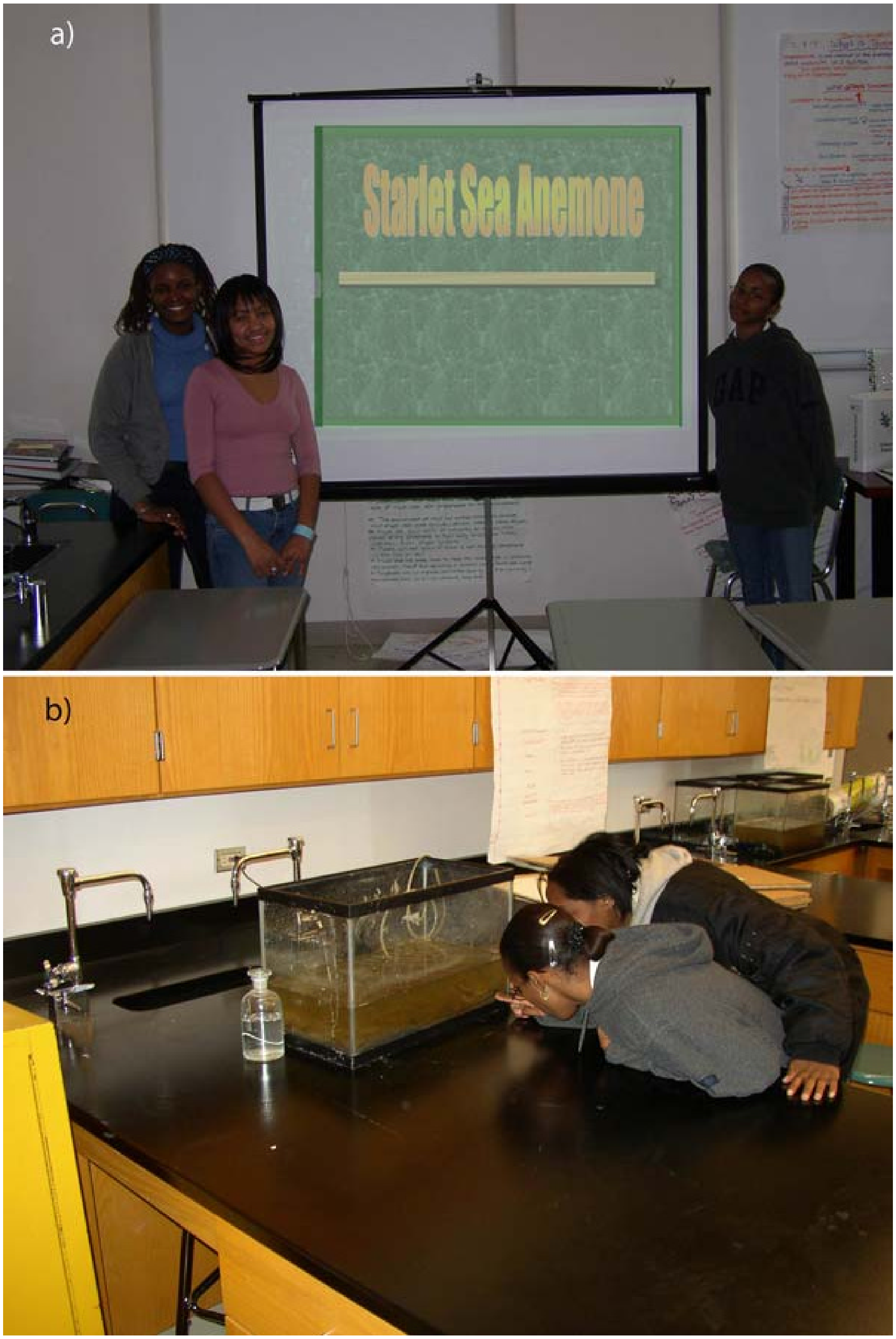

**Figure 3. Terrestrial (a) and marine coral reef (b) mesocosms**

**Figure 4.** **Sample of pilot presentation done using PowerPoint software (Odyssey High School, South Boston, MA, 2004)**

Starlet Sea Anemone

**Table 1. The ten largest urban areas worldwide. Urban areas located on estuaries are in boldface type (Park, 2005).**

| Rank | Urban Area | Population in millions (1999) |
|---|---|---|
| **1** | **Tokyo, Japan** | **28.8** |
| 2 | Mexico City, Mexico | 17.8 |
| 3 | Sao Paulo, Brazil | 17.5 |
| **4** | **Bombay, India** | **17.4** |
| **5** | **New York, United States** | **16.5** |
| **6** | **Shanghai, China** | **14.0** |
| 7 | Los Angles, United States | 13.0 |
| 8 | Lagos, Nigeria | 12.8 |
| **9** | **Calcutta, India** | **12.7** |
| **10** | **Buenos Aires, Argentina** | **12.3** |

**Table 2. Racial demographics of Odyssey High School, Boston, MA, during the 2004–05 year**

| Grade | Black | White | Asian | Hispanic | Native American | Total | Row % |
|---|---|---|---|---|---|---|---|
| **9** | 81 | 36 | 7 | 58 | 0 | 182 | 41.8% |
| **10** | 42 | 20 | 11 | 34 | 0 | 107 | 24.6% |
| **11** | 37 | 25 | 5 | 15 | 0 | 82 | 18.9% |
| **12** | 31 | 11 | 9 | 13 | 0 | 64 | 14.7% |
| **Total** | 191 | 92 | 32 | 120 | 0 | 435 | |
| **Column %** | 43.9% | 21.1% | 7.4% | 27.6% | 0.0% | | |

# Appendix A. National Science Education Standards

This appendix indicates that this lesson meets the National Science Education Standards (National Research Council, 1996). Italicized text from *Inquiry and the National Science Education Standards* (National Research Council, 2000) is followed by a plain text description of how this lesson plan supports the specific standard.

**Standard A—Teachers of science plan an inquiry-based program for their students. In doing this, teachers**

- *Select science content and adapt and design curricula to meet the interests, knowledge, understanding, abilities, and experiences of students.*

By teaching general concepts of biology and environmental science through urban organisms and ecosystems, the lesson utilizes preexisting knowledge of the students' surroundings. By requiring students to conduct observational and literature research, the lesson utilizes the abilities of the students and leads to understanding and knowledge of the lesson content.

- *Select teaching and assessment strategies that support the development of student understanding and nurture a community of science learners.*

The students are given an excellent starting point for their independent work through the prepared fact sheets (Appendices C–F). By requiring independent research to complete the performance objective, teaching and assessment strategies are utilized that will greatly enhance student understanding. By requiring groups that have become student experts on one organism to teach other students during presentations, a community of science learners is developed.

**Standard B—Teachers of science guide and facilitate learning. In doing this, teachers**

- *Focus and support inquiries while interacting with students.*

This lesson enables students to be the primary scientific investigators through their own data collection and literature reviews. In this inquiry, teachers play a supportive role working alongside, not in front of, the students.

- *Challenge students to accept and share responsibility for their own learning.*

In addition to questions posed by the instructor(s), students are expected to develop their own questions to research about the organisms. In addition, because students are expected to peer teach the rest of the class on the group's independent work, each group is held responsible by the rest of the class.

- *Recognize and respond to student diversity and encourage all students to participate fully in science learning.*

This lesson plan is written for classes with a wide range of skill levels. The observational and group nature of the lesson plan encourages all students to participate; for example, students who might have strong observational skills but have difficulty conducting a literature review may be aided by other students who might have poor observational skills yet strong research skills.

**Standard C—Teachers of science engage in ongoing assessment of their teaching and of student learning. In doing this, teachers**

- *Use multiple methods and systematically gather data about student understanding and ability.*
- *Guide students in self-assessment.*
- *Use student data, observations of teaching, and interactions with colleagues to reflect on and improve teaching practice.*
- *Use student data, observations of teaching, and interactions with colleagues to report student achievement and opportunities to learn to students, teachers, parents, policymakers, and the general public.*

As part of the lesson, the instructor(s) is expected to continuously monitor and probe student progress through questioning. Furthermore, the culminating presentation, the use of self-assessment or peer assessment, and the use of a homework sheet to integrate concepts allows instructors to utilize a wide range of assessment strategies. Completion of the performance objective also creates a result that can be archived electronically to allow instructors to refine the lesson plan to better serve their needs.

**Standard D—Teachers of science design and manage learning environments that provide students with the time, space, and resources needed for learning science. In doing this, teachers**

- *Structure the time available so that students are able to engage in extended investigations.*
- *Create a setting for student work that is flexible and supportive of science inquiry.*
- *Make available science tools, materials, media, and technological resources accessible to students.*
- *Identify and use resources outside the school.*
- *Engage students in designing the learning environment.*

The inquiry-based and student-designed nature of the project fulfills this national standard. By allowing groups to organize their own time between the tasks of observation, literature review, and presentation development, the lesson "create(s) a setting for student work that is flexible and supportive of science inquiry." By including as part of the fact sheets (Appendices C–F) Internet websites as resources and providing the students with computers that they can use to search those sites, technological science tools and outside resources are made available to the students. By asking that students design as a class the sampling procedure, and allowing individual discretion as to pertinent topics to be covered in group presentations, the students are engaged "in designing the learning environment."

**Standard E—Teachers of science develop communities of science learners that reflect the intellectual rigor of science inquiry and the attitudes and social values conducive to science learning. In doing this, teachers**

- *Display and demand respect for the diverse ideas, skills, and experiences of all students.*
- *Nurture collaboration among students.*
- *Model and emphasize the skills, attitudes, and values of scientific inquiry.*

This lesson relies heavily upon group work conducted by the students. As such, they must be collaborative. Within groups, each student is likely to posses a different "competitively advantageous" skill. That is, since the project requires observational work, literature review, and technical proficiency with Internet exploration and Microsoft PowerPoint software, the lesson "display(s) and demand(s) respect for diverse ideas, skills, and experiences of all students." By necessitating collaboration and rewarding a diverse skill set, the lesson also emphasizes the "attitudes and values of scientific inquiry."

**The Essential Features of Classroom Inquiry**

- *Learners are engaged by scientifically oriented questions:*
- *Learners give priority to evidence, which allows them to develop and evaluate explanations that address scientifically oriented questions.*
- *Learners formulate explanations from evidence to address scientifically oriented questions.*
- *Learners evaluate their explanations in light of alternative explanations, particularly those reflecting scientific understanding.*
- *Learners communicate and justify their proposed explanations.*

In this lesson, students are engaged by scientifically oriented questions posed by the instructor(s) as well as by each other during their observation. They collect their own evidence to answer some of these questions during observation and perform literature reviews to research other questions. As per the performance objective, students are required to communicate and justify their research results through an oral presentation.

# Appendix B. Project Introduction Sheet

**Mesocosm Project**

**Purpose**

The purpose of this project is to collect and examine sediment and organisms from an urban salt marsh and learn about these organisms and their place in their ecosystem.

**Objective**

At the conclusion of this project, each group will present information in the form of a PowerPoint presentation to the other groups regarding one of the following:

a. The Banded Killifish

b. The Starlet Sea Anemone

c. The Periwinkle Snail

d. The Grass Shrimp

Information will be provided via fact sheets containing information and Internet-based resources. Students are encouraged to research their organism using the Internet.

**Presentation Format**

The PowerPoint presentation will consist of between 7 and 10 slides and take the following format:

a. Title slide with group members' names

b. 3 slides containing information about your organism: These slides should include information about the geographic range of your organism, its habitat, its life cycle, what it eats, what eats it, and any other information you deem important.

c. Between 1 and 3 slides of your observations: These slides should include any observations that you make about your organism (how it swims, how it eats, etc).

d. 3 slides about your organism and urban ecology: These slides should include information about how your organism is affected by humans and how your organism contributes to humans.

# Appendix C. Fundulus heteroclitus Fact Sheet (to be handed out to students)

Note: For the purposes of uninterrupted reading by students, references are not placed in the body of the fact sheet. References may be found in Literature Cited.

## The Banded Killifish (Mummichog)

### General Info and Habitat

The mummichog, *Fundulus heteroclitus*, is an estuarine species that can tolerate a wide range of temperatures and salinities. It often coexists with another *Fundulus* species, *Fundulus diaphanus*, and is often found in schools of large numbers. Growing up to 5 inches, when found in salt pools, they may be the largest predators; but in tidal creeks and open bays of estuaries, they are preyed upon by many larger fish.

### What They Eat

The mummichog is an omnivore that consumes a wide range of organisms. It will eat eelgrass fragments, insect larvae, smaller fish, fish eggs, diatoms, mollusks, crustaceans, and worms in the sediment.

### What Eats Them

The mummichog is an important food source for larger fish and wading birds.

### Range

The mummichog is found from the Atlantic coast of Florida north through the Gulf of St. Lawrence in Canada.

### Life Cycle

The mummichog spawns between April and August. Spawning occurs at high tides with the new and full moons, usually at night. Clutches of eggs, which may number up to 300, are deposited in mussel shells, on the underside of eelgrass leaves, or in other hidden places where they are protected from drying even after the high tide recedes. The eggs hatch after the tides reach them again.

### The Urban Connection

This small fish is particularly well adapted to survive in urban environments. PCB's (polychlorinated biphenyls) are a type of contaminant that can prevent many organisms from surviving and reproducing. *Heritable altered gene expression* in populations of fish from polluted areas is thought to aide these fish in their survival in heavily polluted estuaries. Because of this genetic change, these fish have been found in heavily polluted harbors such as New Bedford, MA, and Newark Bay, NJ.

The mummichog has also been used in a wide range of scientific studies, because it is a *vertebrate* that is capable of being held in large populations for experimental studies in laboratories. It has been an experimental animal used to study evolution, toxicology, and endocrinology.

But this fish's value to people extends beyond the laboratory. The mummichog is an important food source for larger fish and is important in maintaining their populations. Many fishermen use the mummichog as bait while fishing. And mummichogs are also important for mosquito control. They have even been introduced into ponds and ditches because they eat the mosquito larvae. One mummichog may eat as many as 2,000 larvae a day! This is important not only to prevent urban dwellers (like us) from itching annoying mosquito bites, but may even help prevent the spread of dangerous diseases spread by the mosquitoes, such as eastern equine encephalitis and West Nile virus.

### Look It Up!

If you didn't understand something you read above, look it up! A "Google" search is a good place to start (www.google.com). Another good place to start is at the American Killifish Association (www.aka.org). It lists a number of "links" that you can use to navigate.

# Appendix D. Nemostella vectensis Fact Sheet (to be handed out to students)

Note: For the purposes of uninterrupted reading by students, references are not placed in the body of the fact sheet. References may be found in Literature Cited.

## The Starlet Sea Anemone

### General Info and Habitat

The starlet sea anemone, *Nematostella vectensis*, is a small anemone that inhabits estuaries. It burrows in soft sediment with only its mouth and tentacles above the surface. It lives in tidal creeks, salt pools, and mud flats. It can tolerate a wide range of temperatures and salinities and has been found in salinities between 2 ppt (nearly fresh water) and 52 ppt (ocean water = 33 ppt). Sea anemones are members of a *phylum* called 'Cnidaria.' Other cnidarians include jellyfish and corals. These are relatively simple animals which possess only two body layers (*diploblasty*).

### What They Eat

The anemone has been found to feed upon insect larvae, copepods, chironomid larvae, snails, worms, and almost anything else it can grab with its tentacles. It burrows in the soft sediment, and when anything small contacts its tentacles, the anemone reacts very quickly, drawing the prey toward its mouth.

### What Eats Them

Nobody truly knows, but it seems that nothing in the salt marsh eats this small anemone. The anemone is very difficult to find, burrowed into the soft sediment. Additionally, this anemone has stinging cells throughout its body. When attacked by a potential predator (like a killifish or grass shrimp), it releases these stinging cells in self-defense!

### Range

The starlet sea anemone ranges from Nova Scotia (Canada) as far south as the Gulf of Mexico. It is also found on the Pacific coast of the United States and in England!

### Life Cycle

Like all anemones, the starlet sea anemone can reproduce either *sexually* or *asexually*. In sexual reproduction, females release eggs, which are fertilized by sperm released by males. Fertilized eggs mature into larvae, which change into adult anemones. In asexual reproduction, an adult anemone divides itself into two halves. Each half regenerates the other half of its body. This process is called transverse fission.

### The Urban Connection

Like the killifish, the starlet sea anemone has been used in many scientific studies. Because it is a simple organism and easy to maintain in the laboratory, scientists have used it to study development and evolution. It has been used to study how complex body forms evolved; how *bilateral symmetry* evolved from *radial symmetry*.

The fact that this sea anemone can reproduce asexually might allow it to better survive in *fragmented* habitats. In impounded marshes and isolated salt pools, populations of sea anemones have been found that consist of one clonal line!

In Britain, where the sea anemone is found in only a handful of places, it is protected by the government. This limits the commercial development of marshes where the anemone is located.

### Look It Up!

If you didn't understand something you read above, look it up! A "Google" search is a good place to start (www.google.com). A good website to find information about the starlet sea anemone is www.nematostella.org.

# Appendix E: Palaemonetes pugio Fact Sheet (to be handed out to students)

Note: For the purposes of uninterrupted reading by students, references are not placed in the body of the fact sheet. References may be found in Literature Cited.

## Grass Shrimp

### General Info and Habitat

The grass shrimp (*Palaemonetes pugio*) is a small translucent organism inhabiting shallow water in estuaries. This clear shrimp is a crustacean with a well-developed serrated anatomical feature known as a "rostrum," which looks like a horn and extends over the eyes. The front two legs of the shrimp have well-developed claws, which it uses for feeding purposes. The shrimp is an excellent and very fast "swimmer." Like many other inhabitants of estuaries, the shrimp can tolerate a wide range of temperatures and salinities, although it generally avoids the high-salinity waters near the ocean.

### What They Eat

Larvae of the grass shrimp feed upon *zooplankton*, *algae*, and detritus. As adults, the shrimp consume a wide variety of food items, including algae, detritus, worms, and crustaceans.

### What Eats Them

The grass shrimp is consumed by fish in great numbers. Killifish, as well as commercially important fish such as sea trout, perch, and bass, take advantage of the large numbers of grass shrimp found in estuaries.

### Range

Massachusetts is the northern limit of the range of the grass shrimp. It may be found anywhere on the Atlantic seaboard south of Massachusetts through the Gulf of Mexico.

### Life Cycle

Grass shrimp spawn in the summer, usually within 10 hours of *molting*. Males fertilize the eggs externally with a *spermatophore* as they emerge from the female. The females then carry the eggs in a brood pouch until they are ready to hatch, after 12 to 60 days, depending upon the temperature. After hatching, the larvae undergo 11 juvenile stages before becoming adults!

### The Urban Connection

Grass shrimp have very little intrinsic economic and recreational value, although some fishermen do use these shrimp as bait. They do serve an important ecological role, however, by breaking down dead plants and animals (detritus).

Additionally, grass shrimp serve as food for commercially important fish species such as sea trout, perch, and bass.

The grass shrimp has been found in polluted waters (e.g., New Bedford, MA, and Charleston Harbor, SC). Research is currently being conducted on how this organism survives there. It appears that a genetic change has allowed this species to survive in areas contaminated by heavy metals, such as *chromium.*

### Look It Up!

If you didn't understand something you read above, look it up! A "Google" search is a good place to start (www.google.com). A good website to find information about the grass shrimp is www.chesapeakebay.net/info/palaemonetes.cfm.

# Appendix F: Littorina Species Fact Sheet (to be handed out to students)

Note: For the purposes of uninterrupted reading by students, references are not placed in the body of the fact sheet. References may be found in Literature Cited.

## Periwinkles

### General Info and Habitat

Periwinkle refers to a group of snails belonging to the genus *Littorina* and includes the following species: the edible periwinkle (*L. littorea, L. littoralis*), the southern periwinkle (*L. angulifera*), the smooth periwinkle (*L. obtusata*), and the gulf periwinkle (*L. irrorata*). Periwinkles are very common inhabitants of salt marshes, estuaries, and intertidal zones. They are a marine snail and have gills that allow them to breathe underwater. They have a spiraled shell, and the larger species can grow up to 1 inch.

### What They Eat

Periwinkles are herbivores, eating algae in the salt marsh. They will also eat cordgrass (*Spartina* species), which have a vital role in supporting the ecosystem of the marsh. They play an important role in breaking down the grasses that have died in the marsh.

### What Eats Them

Periwinkles are often present in great abundance. Crabs, fish, birds, and small mammals in the salt marsh all feed upon them.

### Geographical Range

Periwinkles are found along the Atlantic coast from Nova Scotia, Canada through the Gulf of Mexico. The edible periwinkle is actually *endemic* to Europe and was introduced to North America.

### Life Cycle

Periwinkles reproduce sexually and may reproduce all throughout the year in parts of their range. Mating peaks in late spring or early summer. Fertilization occurs internally. The white, oval-shaped egg masses may contain over 250 eggs and hatch after four weeks. The larvae grow and metamorphose into sexually mature snails after two years. An interesting aspect of the life cycle of these snails is that they can change their sex!

### The Urban Connection

Periwinkles play an important role in the ecosystem by breaking down dead plant material. They are sensitive to many types of contamination. Some pollutants affect the ability of the snail to reproduce, while others can change its sex organs! Because of this, scientists use them to study the affects of pollution upon salt marsh species.

Periwinkles are an important food source, not only for organisms like crabs and birds, but for people too! The edible periwinkle is so called because they can be eaten. Europeans have always made use of this commonly abundant food source, but it is not commonly eaten in North America.

Its use as a food source may be the reason why the edible periwinkle was introduced into North America. It first appeared off of the coasts of Nova Scotia and New Brunswick, Canada, in the 1850's. Since then, this very successful species has spread south through Florida and the Gulf of Mexico.

But the introduction of a new species is not always a good thing. Introduced species can disrupt the delicate balance of an ecosystem. A recent scientific study addressed what would happen if crabs, which are commercially harvested, were completely removed from Atlantic marshes. Without the crabs to eat the periwinkles, the periwinkle populations may grow out of control. Unchecked, these populations of periwinkles can devastate the cordgrass (*Spartina*) that forms the base of the salt marsh ecosystem. Without the cordgrass, all of the animals that rely on this highly productive plant cannot survive in the marsh either.

### Look It Up!

If you didn't understand something you read above, look it up! A "Google" search is a good place to start (www.google.com). A good website to find information about the periwinkle is www.marlin.ac.uk/species/Litobt.htm. Try to find a recipe that uses periwinkles

# Benthic Communities in *Spartina alterniflora*– and *Phragmites australis*– Dominated Salt Marshes in the Hackensack Meadowlands, New Jersey*

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## Abstract

*Phragmites australis* is invasive in Atlantic coastal salt marshes and often replaces the native cordgrass, *Spartina alterniflora*. Our research focused on benthic communities found in natural and mitigated *P. australis* and *S. alterniflora* salt marshes at two sites, Sawmill Creek (natural sites) and Mill Creek (mitigation sites) in the Hackensack Meadowlands, New Jersey. The area sampled at the natural sites consisted of adjacent stands of *P. australis* (P) and *S. alterniflora* (S) separated by a tidal creek. At Mill Creek, samples were collected at a 12-year-old mitigation site consisting of *S. alterniflora* (M-12) and a newly mitigated site (M-0) where site regrading had just taken place. Benthic samples were taken at the creek bank and the edge of vegetation for all sites. Data on salinity levels and textural and structural sediment characteristics were collected at each site. Our data indicate that both the *P. australis* and *S. alterniflora* sites support diverse benthic communities, although there were differences in diversity and composition in the communities found among the different types of grasses. The average abundance of benthic invertebrates at the Sawmill Creek natural sites ranged from about 35,000 per square meter (/m$^2$) to over 240,000/m$^2$. Taxon richness was ~10 at P and ~7 at S and the *P. australis* site had more common taxa present (1.0% of the total abundance). Average abundance of benthic invertebrates at the Mill Creek mitigation sites ranged from about 130,000/m$^2$ to 3 million/m$^2$. Taxon richness was ~10 at M-12 and ~7 at M-0. This study

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suggests that significant change has not occurred at M-12 in the low marsh since the commencement of the mitigation in 1988.

**Keywords:** Benthic invertebrates; estuarine ecology; Hackensack Meadowlands; low marsh; *Phragmites australis*; salt marsh; *Spartina alterniflora*; tidal estuary

## Introduction

*Spartina alterniflora* (salt marsh cordgrass) dominates the low marsh zone of tidal marshes on the East Coast of the U.S. (Teal, 1962; Bertness, 1991). *Spartina alterniflora* provides habitat for salt marsh species, including a food source for benthic invertebrates and insects (Teal 1962; Van Dolah, 1978; Healy & Walters, 1994; Kneib, Newell, & Hermeno, 1997; Able & Hagan, 2000; Graca, Newell, & Kneib, 2000). Another salt marsh grass, *Phragmites australis* (common reed) is an invasive species usually found on high marshes, but it has been moving into low marshes and replacing *S. alterniflora* (Fell et al. 1998; Angradi, Hagan & Able, 2001; Windham & Lathrop, 1999; Weinstein & Balletto, 1999). *Phragmites australis* changes the marsh physically, hydrologically, and chemically (Angradi et al., 2001; Windham & Lathrop, 1999), and this can affect the utilization of the marsh by fish, birds, and other animals. *Phragmites australis* has been replacing native vegetation on the Atlantic and Gulf coasts since the early 1900s (Weinstein & Balletto, 1999; Fell at al., 1998; Angradi et al., 2001; Windham & Lathrop, 1999). Marsh managers have responded by trying to decrease the dominance of *P. australis* on salt marshes. Numerous restoration projects have been undertaken in which *P. australis* was removed and *S. alterniflora* replanted. However, there have been few direct comparisons of the relative level of function of marshes before and after restoration.

Recent studies on *P. australis* have found it to be an ecologically functional habitat for salt marsh inhabitants such as nekton (fishes and swimming decapod crustaceans) and benthic invertebrates (Rilling, Fell, & Warren, 1998; Meyer, Johnson, & Gill, 2001; Angradi et al., 2001; Fell et al., 1998). There have also been recent studies, using stable isotopes, which have found *P. australis* to be a food source for fish (Wainright, Weinstein, Able & Currin, 2000; Weinstein et al., 2000). However, other studies have found *P. australis* to be poor nursery habitat for the mummichog, *Fundulus heteroclitus* (Able and Hagan 2000; Raichel, Able, & Hartman, 2003).

Benthic invertebrates are vital to a functioning salt marsh ecosystem. They are a food source for many salt marsh inhabitants (Kneib, 1988; Fell et al., 1988; Sarda, Foreman & Valiela, 1995) and important components of the estuarine food web (Ishikawa, 1989). Benthic communities may be affected by the invasion of *P. australis*, though studies to determine the nature of the effects have been contradictory. Fell et al. (1998) conducted research along the Connecticut River and found that *P. australis* salt marshes were functionally equivalent to non-*P. australis* salt marshes. Four high-marsh macroinvertebrates (*Orchestia grillus*, *Philoscia vittata*, *Melampus bidentatus*, and *Succinea* species)—all of them prey species for *F. heteroclitus*—were the focus of this study, and they were found in both marsh types. However, Angradi et al. (2001) found that a *Spartina* marsh had greater production of benthic infauna than a *Phragmites* marsh. Overall abundance of benthic invertebrates was higher, and taxon richness was significantly higher in the *Spartina* marsh than the *Phragmites* marsh at all sampling positions and dates. Dominance by the three most abundant taxa (Oligochaeta, Nematoda, and *Manayunkia aestuarina*) was greater in the *Phragmites* marsh (> 85%) than the *Spartina* marsh at most of the sampling positions, indicating a lower benthic diversity in the *Phragmites* marsh (Angradi et al., 2001). Posey, Alphin, Meyer, and Johnson (2003) found only minor differences between *Spartina* and *Phragmites* marshes in the Chesapeake Bay, although most species were slightly more abundant in *Spartina*.

Benthic invertebrates can be used to assess whether or not salt marsh restorations are functioning normally as ecosystems. In order to create a functioning ecosystem, a salt marsh restoration should include the reintroduction of vegetation and the duplication of the nekton and benthos, along with other environmental factors of the marsh (Packard & Stiverson, 1976; Allen et al., 1994; Sacco Seneca & Wentworth, 1994). Studies comparing natural and restored marshes have found similarities and differences with regard to fauna (Minello & Webb, 1997; Minton, 1999; Craft, Broome & Sacco, 1998; Havens, Varnell & Bradshaw, 1995; LaSalle, Landin & Sims, 1991; Sacco et al., 1994; Moy & Levin 1991). However, it’s uncertain how many years it takes for benthic communities in restored or created marshes to become comparable to those in natural marshes. Created marshes are inhabited by opportunistic benthic species, which develop according to the sediment, hydrodynamics, and vegetation of the marsh (Posey, Alphin & Powell, 1997). The more recently created marshes studied by Posey et al. (1997) had more polychaetes than the older created marshes, which had more oligochaetes and amphipods. Packard and Stiverson (1976) suggest that a *Spartina* marsh restored on dredge spoils will eventually sustain a detritus-based community, which is a major food source for benthic invertebrates.

In a few studies, restored salt marshes have been deemed to have characteristics similar to natural marshes (Posey et al., 1997). Some studies indicate that the benthic communities in natural and restored marshes are similar. For example, LaSalle et al (1991) found that benthic communities, along with fish and shellfish, of a natural marsh that developed on dredged material were similar in species composition and abundance to those of natural marshes.

Our study has two objectives. The first is a comparison of the benthic communities of a natural *Spartina* marsh and a *Phragmites* marsh. The second is a comparison of two restored marshes of different ages (> 10 years, < 1 year).

## Methods

### Study Sites

The Hackensack Meadowlands District covers 32 square miles in Bergen and Hudson counties of New Jersey (Figure 1). It is uncertain when *P. australis* first appeared here, but published sightings of the plant in New Jersey date back to the 1800s (Willis, 1877; Britton, 1889; Harshberger & Burns, 1919). In our study, we conducted a comparison of natural marshes dominated by *S. alterniflora* and *P. australis* at Sawmill Creek (natural sites, Figure 1) and an assessment of mitigated intertidal marshes at Mill Creek (mitigation sites, Figure 1).

### Sawmill Creek Natural Sites

The study sites located at Sawmill Creek run along the southern section of the Hackensack River in Lyndhurst, New Jersey. This area is a natural salt marsh that was formed in 1950 after a tide-gate breach (Kraus & Kraus, 1988). The known salinity range at this site is 6.9 to 15.7 parts per thousand (ppt)—mesohaline—and the tidal range is approximately 1.5 meters. *Phragmites australis* is the dominant vegetation at the natural sites, but extensive marshes of *S. alterniflora* can be found as well. We sampled from microhabitats located in adjacent stands of *P. australis* (P) and *S. alterniflora* (S) separated by a tidal creek (Figure 2).

### Mill Creek Mitigation Sites

The mitigated marshes were located at Mill Creek, in Secaucus, New Jersey (Figure 1), an area dominated by *P. australis*. One of the marshes was mitigated in 1988, 12 years before this study, while the other was undergoing mitigation at the time of the sampling. The 12-year-old site (M-12, Figure 3), located along the northern part of the Hackensack River (Hackensack Meadowlands Development Commission, HMDC, 2000), covers 63 acres and was mitigated by Hartz Mountain Company. Mitigation consisted of removing *P. australis*, decreasing site elevation to enhance daily tidal inundation, planting *S. alterniflora*, and site monitoring (TAMS, 1990). This site is completely surrounded by industry, highways, and numerous forms of urbanization (TAMS, 1990; HMDC, 2000). The tidal range at M-12 is approximately 1.5 meters, and the salinity range is 0.5 to 7.0 ppt—oligohaline (Kraus & Kraus, 1988).

The newly mitigated site (M-0, Figure 4), located behind the Mill Creek Mall, in Secaucus, covers an area of 140 acres and was overseen by the New Jersey Meadowlands Commission (formerly the Hackensack Meadowlands Commission). Mitigation began here in 1998 and was conducted to eliminate *P. australis*, increase tidal inundation, and create a low marsh system (HMDC, 2000). At the time of sampling at the M-0 site, the mitigation was still under way, and all that remained was bare substrate, some *P. australis*, and *Pluchea purpurascens*, an annual salt marsh fleabane that flowers between late summer and fall (Newcomb, 1977).

### Organic Matter and Particle Size

We took sediment samples to a depth of 5 centimeters (cm) in September 1999 using a PVC corer. Three replicates were made at each of the four sampling stations. Samples were kept in a cold room prior to processing. The samples were initially sieved wet through a 4.75-millimeter (mm) sieve to collect any pebbles, stones, and large pieces of organic material. We determined the following sediment characteristics: percentage organic matter and percentage silt, sand, and clay.

To determine organic content, five grams of wet sediment were placed in a ceramic crucible, weighed, and placed in a 105°C oven for 16 hours. Samples were then reweighed to determine the dry weight, placed in a hood, and put in a 440°C muffle furnace for 16 hours. Samples were then cooled in the hood and reweighed in order to determine ash dry weight. The percentage of organic matter was calculated using the following equation: (Dry Weight – Ash Weight)/Dry Weight × 100.

To determine particle size, the remaining sediment samples were air dried in a university greenhouse. Pieces greater than 3.175 mm in diameter were removed from the sediment after it was crushed with a mortar and pestle. The sediment was analyzed using the LaMotte Soil Texture Unit (code 1067, LaMotte Co., Chestertown, MD), which provides a volumetric calibration of sand, silt, and clay through sedimentation in an aqueous solution.

### Benthic Sampling

We collected benthic samples in the natural (S and P) and mitigation (M-0 and M-12) sites at two low-marsh microhabitats: (1) the creek bank and (2) the edge of each type of vegetation on the marsh surface. At the natural sites, the creek bank sampled was a steep area coming off the vegetated marsh surface into an emergent area in the intertidal zone. At the mitigation sites, the creek bank sampled was a gently sloping emergent area in the intertidal zone coming off the vegetated (M-12) or nonvegetated (M-0) marsh surface. At M-0, the "edge of vegetation" was estimated by examining vegetation surrounding the site.

Benthic samples were taken at all sites with a 3.9-cm-diameter PVC core sampler. The cores were taken to a depth of 5 cm at each sampling station. Cores were taken at this relatively shallow depth because it has been shown that in stressed marshes (for example, those that are contaminated or have low dissolved oxygen), the large, deep-burrowing invertebrates disappear, and the community becomes dominated by small animals that live close to the sediment surface (Warwick, 1993). Weis, Skurnick, and Weis (2004) have found an absence of larger, deeper-dwelling fauna in the Meadowlands.

We collected three replicates at all sampling sites. Creek-bank samples were collected once a month at low tide from June to September 1999. Edge-of-vegetation samples were collected from July to September 1999. Surface-water salinity at each site was measured with a refractometer.

### Biota

Unsieved benthic samples (mud and organisms) were preserved in 20% formalin in the field. After one to two weeks, samples were transferred to 70% ethanol with rose bengal dye for staining of the benthic invertebrates. The samples were sieved in a 0.3-mm sieve, sorted, and identified in a petri dish to the lowest possible taxonomic level to determine taxa richness, overall abundance, and species composition (Weiss, 1995). The numbers of nematodes were estimated for a site when an average of $\geq 100$ nematodes were found in each sample. The petri dishes utilized were counted and multiplied by the mean number of nematodes in the first two petri dishes sorted for that particular site. For this study, composition comparisons focused on those taxa that were classified common ($\geq 1.0\%$ of the total abundance) and uncommon ($> 0.1\%$ but $< 1.0\%$) (LaSalle & Rozas, 1991).

### Statistical Analysis

We analyzed all the benthic samples using the Statistical Analysis System (SAS) software (Version 8). Multiple analysis of variance (MANOVA) and the Student-Newman-Keuls (SNK) tests were used to analyze the monthly benthic samples ($p < 0.05$). The percentage of organic matter and the percentage of silt, sand, and clay were analyzed by running analysis of variance (ANOVA). All data were tested for normality, and any abundance data with abnormal distribution were log-base-10 transformed. A commonly used measure of diversity, the Shannon-Wiener Index, was calculated using Multivariate Statistical Package (MVSP) software (Version 3.12c).

## Results

### Sawmill Creek Natural Sites

### Organic Matter, Particle-Size Distribution, and Salinity

We found a significant difference between the percentage of organic matter in the creek-bank sediment samples from S (3.08%) and P (1.58%) (Table 1 and Table 2, $p = 0.0003/df = 2$). There was no significant difference in percentage of organic matter at the edge of the vegetation among the natural sites (Table 1 and Table 2, S: 2.18%; P: 2.28%). However, the mean percentage of organic matter was significantly different between the creek bank and the edge of the vegetation for both *P. australis* ($p = 0.0656/df = 1$) and *S. alterniflora* ($p = 0.0251/df = 1$).

We found no significant differences in percentage of silt in the creek-bank samples at the natural sites. The sediment was significantly coarser at S than P on the creek bank since S had more sand present (Table 1 and Table 2, $P = 0.0038/df = 2$). There were fewer fine particles (less clay content) at S than P (Figure 5, Table 2, $p = 0.0128/df = 1$). We found no significant difference between sites in particle-size distribution at the edge of the vegetation.

At both natural sites, the salinity was 8 ppt in June, 15 ppt in July, 20 ppt in August, and 4 ppt in September. The increase in salinity from June to August was due to a drought during the summer of 1999. The drastic salinity decrease in September was due to rainfall associated with Hurricane Floyd on September 16, 1999.

### Abundance, Richness, and Composition

We found no significant differences in the total number of organisms over all months sampled on the creek bank for the natural sites (Figure 6, Table 3 and Table 4, $p > 0.05/df = 1$). Mean abundance at the edge of the vegetation was not significantly different for July and August (Figure 6, Table 4), but during September, P had significantly less abundance than S ($p = 0.0212/df = 1$). The natural sites showed no significant difference for July, August, and September between creek-bank and vegetation-edge habitat types for S or for P.

There was no significant difference in taxa between S and P at the creek bank for all months sampled. The mean taxa richness at the edge of the vegetation did show significant differences for some months (Figure 7, Table 3 and Table 4, $p < 0.05/df = 1$). During August, P had a significantly higher number of taxa than S at the edge of the vegetation (Table 4, $p = 0.0255/df = 1$). However, there was no significant difference between S and P at the creek bank for July and September. In August, the creek bank of S had significantly higher mean taxa richness ($p = 0.0241/df = 1$) compared to P.

Presence (+) and absence (–) of the 25 taxa found over the four months at the sites is documented in Table 5. Three types of meiofauna (Nematoda, Copedoda, and Ostracoda) and several macrofaunal taxa dominated the collections. We identified ten common taxa and eight uncommon taxa at P (Table 6). Eight common taxa were found at S, along with six uncommon taxa (Table 6). Oligochaeta (24.4%) had the highest percent composition at P, while *Manayunkia aestuarina* (33.0%) had the highest percent composition at S.

### Shannon-Wiener Diversity Index

We found a significant difference between taxa diversity on the creek banks of the different sites for June. In this month, P was significantly more diverse than S ($p = 0.0347/df = 1$), but during July, August, and September, there was no significant difference ($p > 0.05/df = 1$) between P and S (Table 3 and Table 4). At the edge of the vegetation, P had significantly more diversity than S in August ($p = 0.0371/df = 1$), but not in July and September ($p > 0.05/df = 1$).

The taxa diversity between the two habitats at S was significantly different in July ($p = 0.0218/df = 1$) and August ($p = 0.0189/df = 1$), when the creek bank was more diverse than the edge of the vegetation (Table 4). For September, there was no significant difference between the two habitats at S. At P, there was no significant difference in July and September ($p > 0.05/df = 1$), but in August the creek bank was significantly more diverse than the edge of the vegetation ($p = 0.0081/df = 1$).

## Mill Creek Mitigation Sites

### Organic Matter, Particle Size Distribution and Salinity

Organic matter at the creek bank for M-0 was calculated to be 1.69% compared to 1.33% at M-12 (Table 1 and Table 2, $p = 0.0027/df = 2$). There was no significant difference at the edge of the vegetation among the sites (M-12: 2.02%; M-0: 2.31%). For both sites, the mean percentage of organic matter was significantly higher at the edge of the vegetation habitat than the creek bank (M-12: $p = 0.0088/df = 1$; M-0: $p = 0.0075/df = 1$).

We found no differences in silt, sand, and clay percentages (Table 1 and Table 2; Figure 8) between the creek banks of both mitigation sites. In addition, no difference was discernable at the edge of the vegetation between the sites.

Salinity at the mitigation sites (M-12) was 3 ppt in June, 10 ppt in July, 10 ppt in August, and 2 ppt in September. The mitigation sites at Mill Creek, following a pattern similar to the natural sites at Sawmill Creek, decreased in salinity in September in concurrence with the rainfall of Hurricane Floyd.

### Abundance, Richness, and Composition

We found no significant difference in the total number of organisms on the creek bank of the mitigation sites for all months sampled (Table 3 and Table 4, Figure 9). The mean abundance at the edge of the vegetation was not significantly different between the sites for July (Figure 9, Table 4, $p = 0.8001/df = 1$). However, during August ($p = 0.0133/df = 1$) and September ($p < 0.0001/df = 1$), abundance at M-12 was significantly greater than at M-0. Abundance was greater at the marsh edge than the creek-bank habitats for July at M-12 ($p = 0.035/df = 1$); and it was greater at the creek bank than the marsh edge for August ($p = 0.0024/df = 1$) and September ($P < 0.0001/df = 1$) at M-0 (Figure 9, Table 4).

In August and September, the taxa richness on the creek bank at M-0 was lower than at M-12 (Table 3 and Table 4, Figure 10, August: $p = 0.0004/df = 1$; September: $p = 0.0058/df = 1$). During July, M-12 had more taxa present than M-0 at the edge of the vegetation (Table 3 and Table 4, Figure 10, $p = 0.0048/df = 1$). The mean taxa richness for M-12 was significantly higher at the creek bank in August ($p = 0.0158/df = 1$) and September ($p = 0.0335/df = 1$) than at the edge of the vegetation.

Refer to Table 5 for presence (+) and absence (–) of the 25 taxa found over four months at all of the sites. At M-12, four common taxa were found along with two uncommon taxa (Table 6). Four common taxa and three uncommon taxa were found at M-0 (Table 6). Nematoda were the most abundant taxa at both M-12 and M-0. The mitigation sites were heavily dominated by nematodes, which comprised approximately 77%–80% of all taxa.

### Shannon-Wiener Diversity Index

In August, we found that M-12 was more diverse than M-0 at the creek bank (Table 3 and Table 4, p = 0.0464/df = 1). During June, July, and September, there was no significant difference between M-12 and M-0. At the edge of the vegetation, M-12 had less diversity than M-0 in August (p = 0.0225/df = 1) and September (p = 0.0014/df = 1), but during July, M-12 was significantly more diverse than M-0 (p = 0.0333/df = 1).

At M-12, benthic taxa diversity was not significantly different between the creek bank and the edge of the vegetation for July, August, or September. The taxa diversity between the two habitats at M-0 was significantly different in August (p = 0.0012/df = 1) and September (p = 0.0007/df = 1), when the edge of the vegetation was more diverse than the creek bank. For July, there was no significant difference between the two habitats at M-0.

## Discussion

### Sawmill Creek Natural Sites

Angradi et al. (2001) compared the benthic communities in *P. australis* and *Spartina* marshes and found that the *Spartina* marsh had a greater abundance and taxa richness than *P. australis* marsh. However, we found no clear pattern of difference in taxa abundance and richness at *P. australis* and *S. alterniflora* marshes during our study period. As in Angradi et al.'s study, oligochaetes, nematodes, and *Manayunkia aestuarina* were the dominant taxa in our study. And both studies included samples from the creek bank and the edge of the vegetated marsh. However, Angradi et al. sampled from within the vegetated zone of the marsh, whereas we did not. Since our samples were collected from the creek bank and the marsh edge only, the data may not be reflective of the marsh surface.

There have been studies that have looked at marsh surface and made comparisons between *P. australis* and *S. alterniflora* marshes, including Posey et al. (2003) and Fell et al. (1998). Posey et al. found that the macrobenthic communities in paired *P. australis* and *S. alterniflora* marshes of Chesapeake Bay—an oligohaline to mesohaline environment—were affected by the vegetation type. While they noted few significant differences in the abundance of most individual taxa, they did find a significant overall community trend toward a higher rank abundance of the invertebrates in the *S. alterniflora* marsh compared to the *P. australis* marsh. Fell et al. looked at epibenthic communities in non-*P. australis* and *P. australis* marshes and found them to be equivalent.

### Mill Creek Mitigation Sites

TAMS (1990) evaluated the benthic community in the open-water channels at M-12 and found it to have low diversity. Ten years after this study, our investigations showed that the low diversity still persists. TAMS also found that the benthic community at M-12 consisted of pollution-tolerant organisms, with oligochaetes and hydrobiid gastropods comprising more than 80% percent of the community. In our study, nematodes made up about 80% of the benthic community, and there were only four common taxa found at this site. We conclude that this site is still dominated by few taxa with large abundances.

Our results are similar to those of Kraus and Kraus (1988), who studied mitigation and natural sites in Sawmill Creek and sampled the mitigated (*S. alterniflora*) and nonmitigated (*P. australis*) sections of Mill Creek. Like us, they found that Sawmill Creek had a greater abundance, higher taxa richness, and higher diversity of benthic invertebrates than the Mill Creek sites, which were dominated by gastropods and nematodes. However, there are two serious obstacles to comparing the studies. First, Kraus and Kraus did not specify the precise locations sampled at Sawmill Creek. Nor did they specify exactly where the Mill Creek control sites were located or which type of vegetation was present there. Second, previous studies have shown that low salinity decreases abundance, taxa richness, and diversity of benthic communities (Levin & Talley, 2000; Boesch, 1972; West & Ambrose, 1992). Insects and oligochaetes usually dominate benthic communities in a low-salinity system, while a high-salinity system is known to favor polychaetes (Levin & Talley). There is evidence for this in our study, which shows that the high-salinity natural sites had considerably more polychaetes than the low-salinity mitigation sites, and that both site types had an abundance of oligochaetes. Oligochaetes were one of the few common taxa found at Mill Creek, while at Sawmill Creek they were one of the many common taxa.

Studies have shown that contaminants and pollutants affect benthic communities (Gray, Clarke, Warwick & Hobbs, 1990; Pocklington & Wells, 1992; Gaston & Young, 1992; Whaley, Garcia & Sy, 1989; Maltby, 1999; Flynn, Wakabara & Tararam, 1998). According to Levin and Talley (2000), a marsh exposed to sewage has a greater abundance of the oligochaete *Monopylephorus rubroniveus* and the amphipod *Talorchestia longicornis*. The mitigation sites at Mill Creek had abundant oligochaetes. A contamination source that may be affecting the benthic community at the mitigation sites is the sewage-treatment plant along Mill Creek. Kraus and Kraus (1988) reported that the water quality was better at the natural sites than at the mitigation sites. However, during the dates of our study, water quality was better at the mitigation sites than at the natural sites (Center for Information Management, Integration and Connectivity, CIMIC, 1999). But water quality on a single date is far less important than long-term sediment concentration, which would impact benthos. Contamination persists in the mitigation sites' sediments, but an experiment replacing contaminated sediments with uncontaminated ones showed no shift in community composition (Yuhas, 2001).

Overall, this study suggests that significant change has not occurred at the M-12 site at the low marsh since the commencement of the mitigation in 1988. Development of a diverse benthic community may not have occurred because of the low salinity that is representative of an oligohaline area. Changing the vegetation from *P. australis* to *S. alterniflora* may not, alone, be able to alter the environmental conditions (Packard & Stiverson, 1976; Allen et al., 1994; Sacco et al., 1994) at Mill Creek that affect the benthic community.

This study was a small pilot effort. It is consistent in several ways with some previous studies of the Hackensack marsh system. However, it also shows that a more in-depth examination of the complex interactions between salinity, vegetation, time, and contamination is needed to understand the structure and dynamics of the benthic community in the system.

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## Glossary

**Amphipod:** A small crustacean of the order Amphipoda.

**Analysis of variance (ANOVA):** Statistical method that yields values that can be tested to determine whether a significant relation exists between variables.

**Benthic:** Organisms (e.g., protozoa, nematodes) living in sediments on sea or lake bottoms.

**Decapod:** A ten-legged crustacean of the order Decapoda (e.g., crab, lobster).

**df (degrees of freedom):** In statistics, degrees of freedom are the number of values in probability distributions that are free to be varied.

**Epibenthic:** Organisms living on the surface of sediments on sea or lake bottoms.

**Infauna:** Benthic organisms (see above) that dig into the sediment bed or construct tubes or burrows.

**Log-base-10 transformed (logarithmic transformation):** Statistical technique whereby each observation or raw value of data is replaced by the log of itself.

**Meiofauna:** Organisms that are larger than microfauna and smaller than macrofauna. They will generally pass through a 1-mm mesh but not a 0.3-mm mesh.

**Mesohaline:** Of or relating to a body of water with a salinity measure between 5 and 18 parts per thousand.

**Mitigation (Mitigation banking):** The process of preserving, enhancing, restoring, or creating habitat to compensate for (current or future) habitat disturbances elsewhere, especially due to development.

**Multiple analysis of variance (MANOVA):** An extension of ANOVA (see above) to cover cases where there is more than one dependent variable.

**Nekton:** Actively swimming marine or freshwater organisms.

**Oligochaetes:** Any of various annelid worms of the class Oligochaeta, including the earthworms and a few small freshwater forms.

**Oligohaline:** Of or relating to a body of water with a salinity measure of less than 5 parts per thousand (or 5 grams of salt per liter).

**$p < 0.05$:** An indicator of statistical significance in which the probability of the result of a study being a chance occurrence is less than 5 in 100.

**Polychaetes:** Any of various annelid worms of the class Polychaeta, including mostly marine worms such as the lugworm, characterized by fleshy-paired appendages tipped with bristles on each body segment.

**Refractometer:** An instrument that measures the composition of liquids using light refraction.

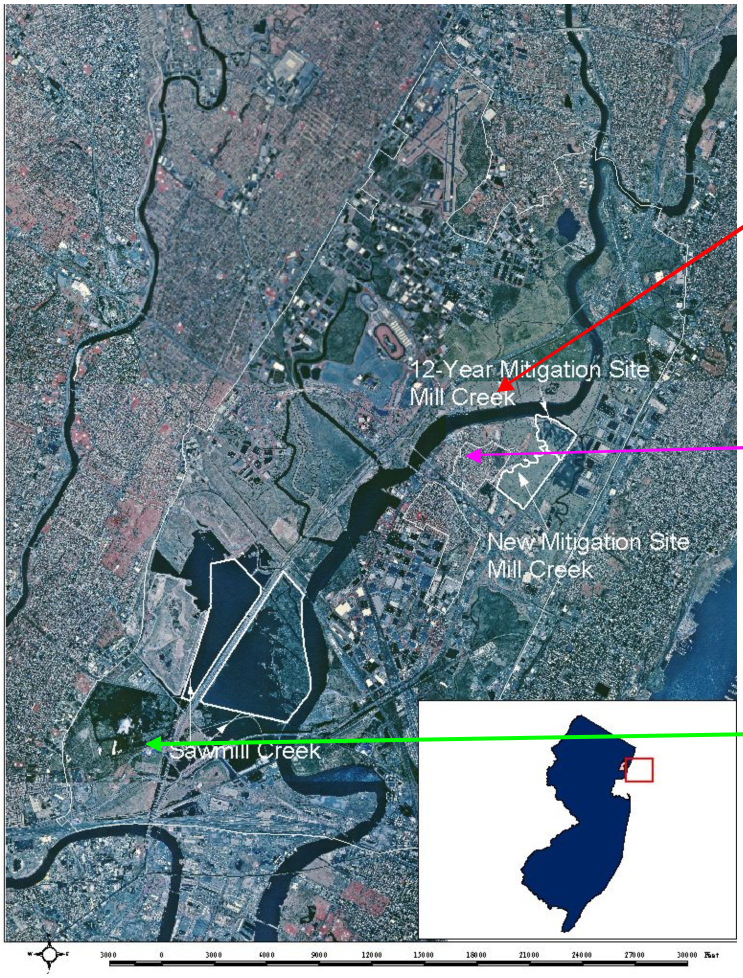
**Richness:** The number of species in an area.

**Stable isotope:** Any naturally occurring, nondecaying isotope (see above) of an element. Many elements have several stable isotopes. For example, carbon (C) has carbon 12 (12C) and carbon 13 (13C).

**Student-Newman-Keuls (SNK) multiple comparison test:** A statistical method for determining differences among groups of samples.

**Taxon (plural: taxa):** A taxonomic rank, such as family, genus, or species.

**Figure 1. Site map of Hackensack Meadowlands District, New Jersey**



**Study Sites**

- **Mill Creek**
  - 12-Year-Old Mitigation Site (Hartz Mountain Mitigation) **(M-12)**
    - *Spartina alterniflora* salt marsh
  - New Mitigation Site
    - Mitigation was in progress at the time of sampling **(M-0)**
      - bare substrate
- **Sawmill Creek**
  - Mature Site
    - Adjacent *Phragmites* **(P)** and *Spartina alterniflora* **(S)** salt marshes

**Figure 2. Sawmill Creek—adjacent stands of *Phragmites australis* (P) and *Spartina alterniflora* (S).**

**Figure 3. Mill Creek—12-year-old mitigation site (M-12).**

**Figure 4.: Mill Creek—new mitigation site (M-0).**

**Figure 5. Mean percentage of silt (horizontal lines), sand (dotted pattern), and clay (vertical lines) at the creek bank and edge of the vegetation habitats of the Sawmill Creek natural sites. Samples were taken at Sawmill Creek—*Spartina alterniflora* (S) and Sawmill Creek—*Phragmites australis* (P) during September 1999.**

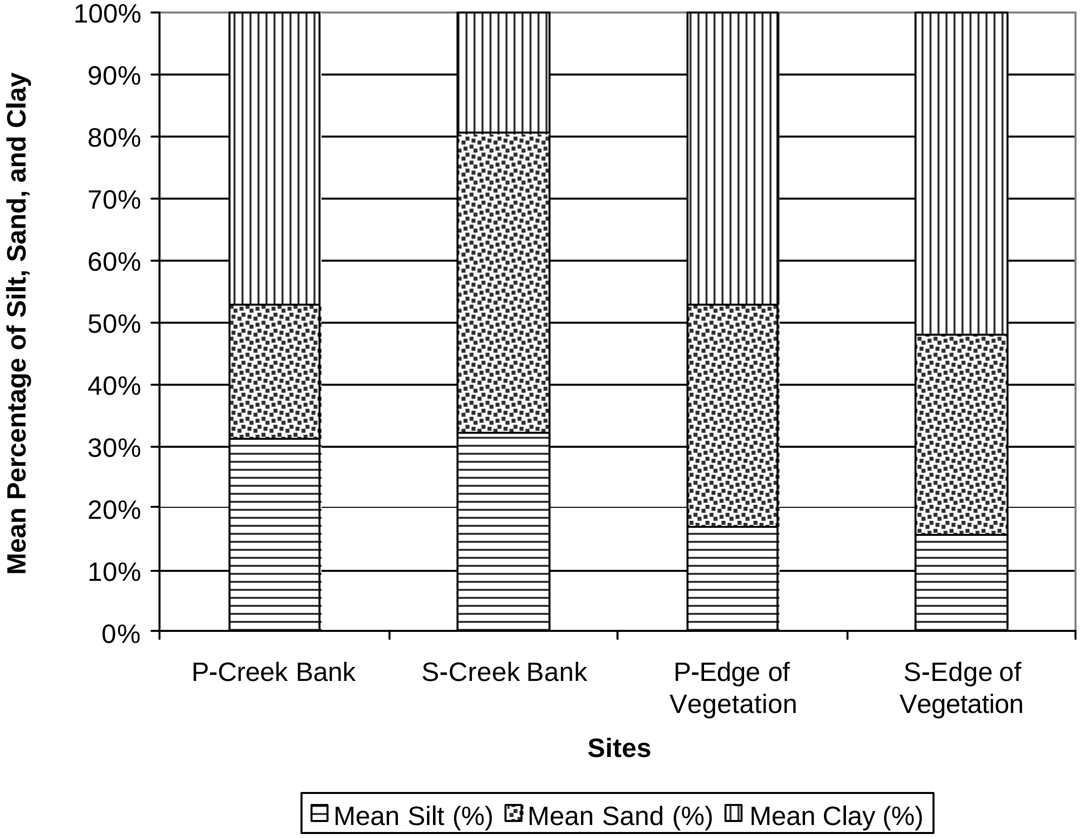


**Figure 6. Mean abundance (#/m²) of benthic invertebrates at the creek-bank habitat and the edge of the vegetation habitats. Sampled at Sawmill Creek—*S. alterniflora* (S) and Sawmill Creek—*P. australis* (P) from June to September 1999.**

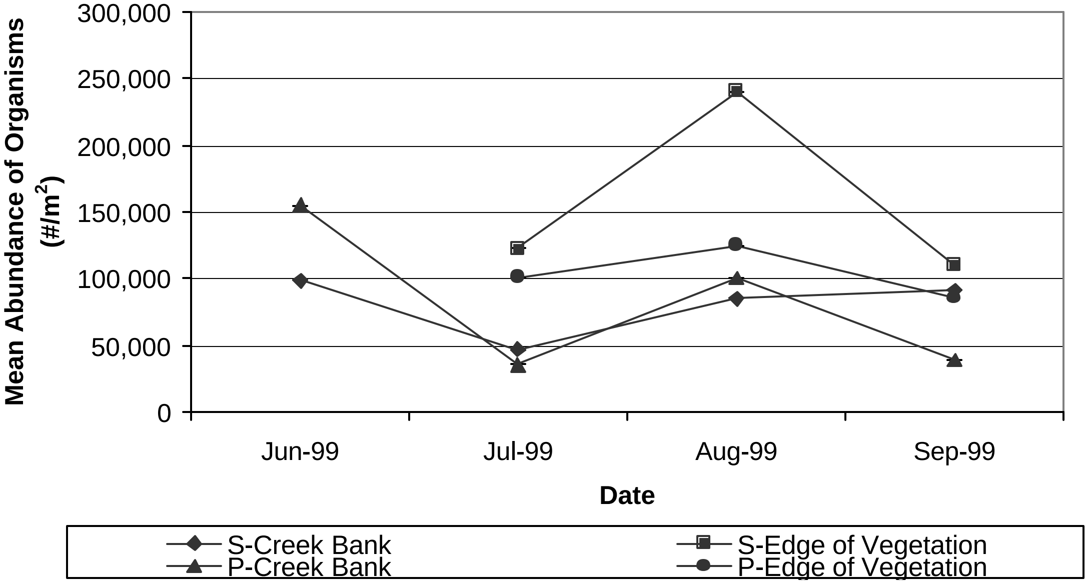


**Figure 7. Mean taxa richness at the creek bank and the edge of vegetation habitats. Samples were taken at Sawmill Creek—*S. alterniflora* (S) and Sawmill Creek—*P. australis* (P) from June to September 1999.**

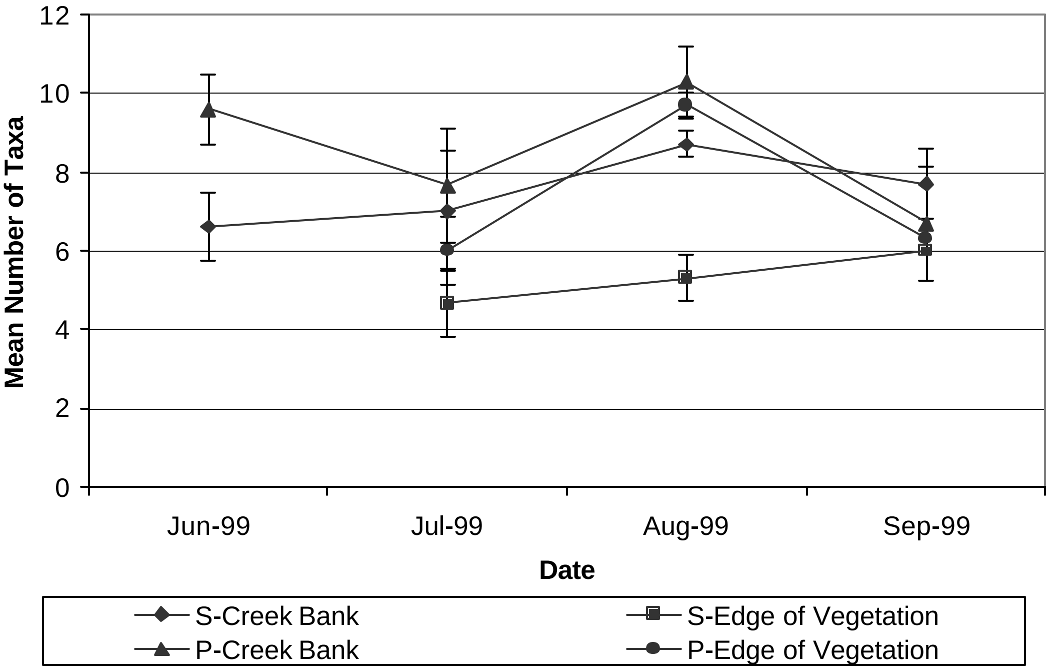


**Figure 8. Mean percentage of silt, sand, and clay at the Mill Creek mitigation sites at the creek bank and edge of the vegetation habitats. Samples were taken at the 12-year-old mitigation site (M-12) and new mitigation site (M-0) during September 1999.**

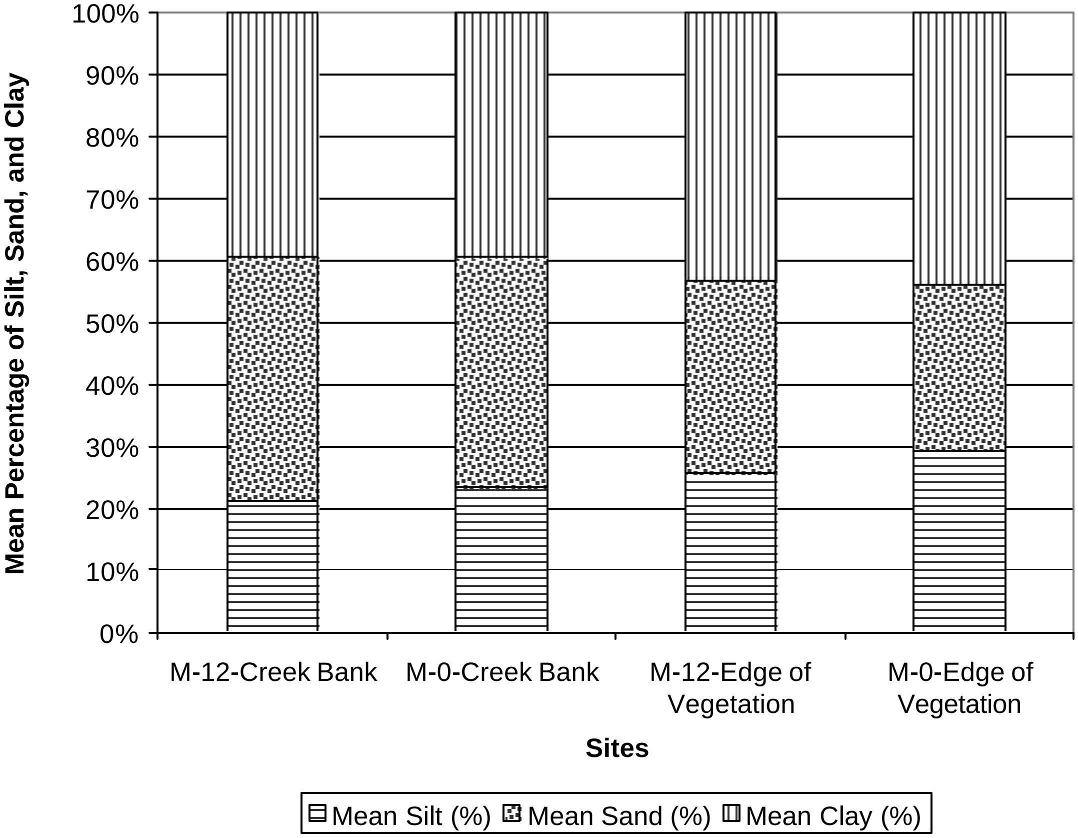


**Figure 9. Mean abundance (#/m²) of benthic invertebrates between the creek bank and the edge of the vegetation habitats. Samples were taken at the 12-year-old mitigation site (M-12) and new mitigation site (M-0) from June to September 1999.**

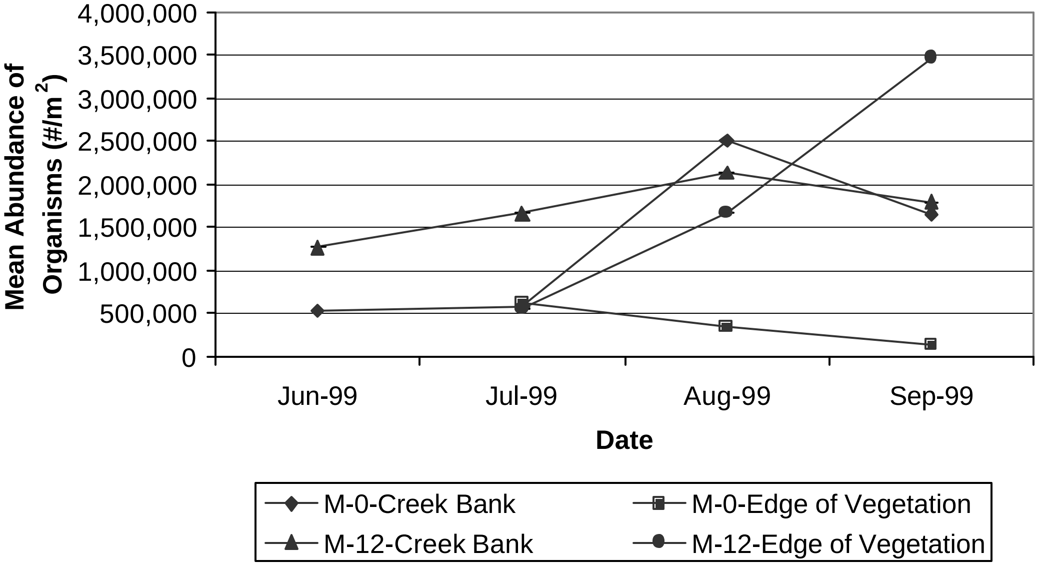


**Figure 10. Mean taxa richness at the creek bank and edge of vegetation habitats. Samples were taken at the 12-year-old mitigation site (M-12) and new mitigation site (M-0) from June to September 1999.**

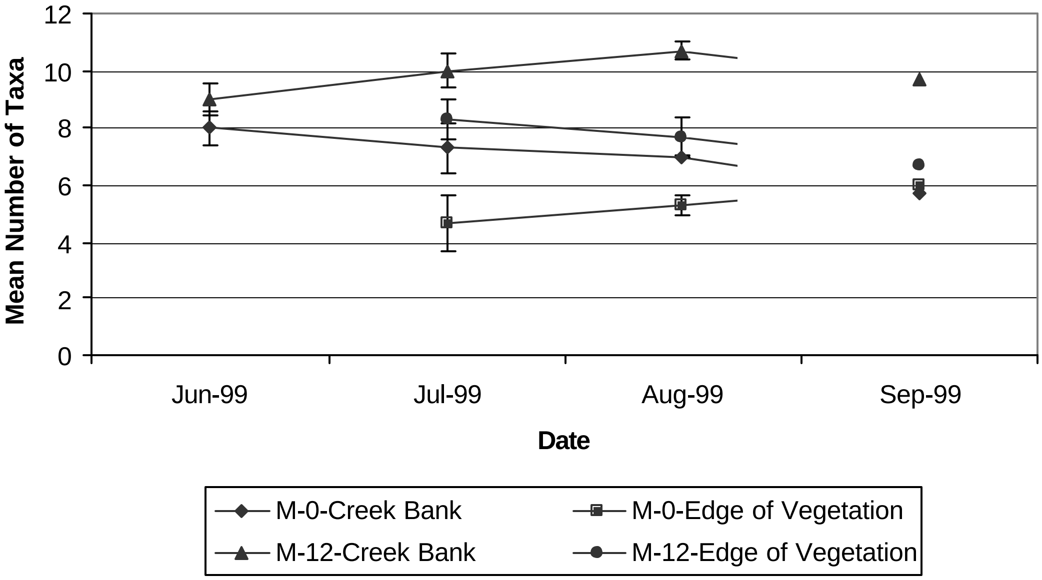


**Table 1. Mean percentage of organic matter, silt, sand, and clay at Sawmill Creek—*P. australis* (P), Sawmill Creek—*S. alterniflora* (S), 12-year-old mitigation site—Mill Creek (M-12), and new mitigation site—Mill Creek (M-0).**

| Site | Mean Percentage of Organic Matter +/- Standard Error | Mean Percentage of Silt +/- Standard Error | Mean Percentage of Sand +/- Standard Error | Mean Percentage of Clay +/- Standard Error |
|---|---|---|---|---|
| P-Creek Bank | 1.582+/-0.07 | 31.1+/-1.1 | 21.8+/-3.63 | 47.1+/-2.57 |
| P-Edge of Vegetation | 2.281+/-0.04 | 17.1+/-0.2 | 35.8+/-2.47 | 47.1+/-2.56 |
| S-Creek Bank | 3.084+/-0.25 | 32.2+/-4.0 | 48.2+/-7.14 | 19.5+/-6.89 |
| S-Edge of Vegetation | 2.157+/-0.08 | 15.8+/-2.7 | 32.2+/-5.53 | 52.0+/-3.16 |
| M-12-Creek Bank | 1.332+/-0.02 | 21.1+/-1.1 | 39.6+/-4.25 | 39.3+/-3.31 |
| M-12-Edge of Vegetation | 2.016+/-0.14 | 25.6+/-11.0 | 31.1+/-9.10 | 43.3+/-5.07 |
| M-0-Creek Bank | 1.688+/-0.09 | 23.3+/-1.2 | 37.1+/-2.32 | 39.5+/-3.47 |
| M-0-Edge of Vegetation | 2.312+/-0.08 | 29.3+/-0.7 | 26.7+/-3.84 | 44.0+/-4.43 |

**Table 2. P values and degrees of freedom for organic matter, silt, sand, and clay at Sawmill Creek—*P. australis* (P), Sawmill Creek—*S. alterniflora* (S), 12-year-old mitigation site—Mill Creek (M-12), and new mitigation site—Mill Creek (M-0).**

| Site | Organic Matter | Silt | Sand | Clay |
|---|---|---|---|---|
| P-Creek Bank<br>P-Edge of Vegetation | P=0.0656/df=1 | P=0.0002/df=1 | P=0.0047/df=1 | P=1.0000/df=1 |
| S-Creek Bank<br>S-Edge of Vegetation | P=0.0251/df=1 | P=0.0273/df=1 | P=0.6548/df=1 | P=0.0128/df=1 |
| P-Creek Bank<br>S-Creek Bank | P=0.0003/df=2 | P=0.5880/df=2 | P=0.0038/df=2 | P=0.0051/df=2 |
| P-Edge of Vegetation<br>S-Edge of Vegetation | P=0.7685/df=1 | P=0.6587/df=1 | P=0.2943/df=1 | P=0.2943/df=1 |
| M-12-Creek Bank<br>M-12-Edge of Vegetation | P=0.0088/df=1 | P=0.7057/df=1 | P=0.5997/df=1 | P=0.5451/df=1 |
| M-0-Creek Bank<br>M-0-Edge of Vegetation | P=0.0075/df=1 | P=0.0106/df=1 | P=0.2397/df=1 | P=0.4719/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.0027/df=2 | P=0.0177/df=2 | P=0.0042/df=2 | P=0.0026/df=2 |
| M-12-Edge of Vegetation<br>M-0-Edge of Vegetation | P=0.1468/df=1 | P=0.7487/df=1 | P=0.9259/df=1 | P=0.9259/df=1 |

**Table 3. Mean abundance (#/m²), mean taxa richness, and mean Shannon-Wiener Diversity Index at Sawmill Creek—*P. australis* (P), Sawmill Creek—*S. alterniflora* (S), 12-year-old mitigation site—Mill Creek (M-12), and new mitigation site—Mill Creek (M-0).**

| **Site** | **Mean Abundance +/- Standard Error (#/$m^2$)** | **Mean Taxa Richness +/- Standard Error** | **Mean Shannon-Wiener Diversity Index +/- Standard Error** |
|---|---|---|---|
| **June** | | | |
| P-Creek Bank | 155030+/-15.52 | 9.67+/-0.88 | 1.66+/-0.06 |
| S-Creek Bank | 98297+/-26.87 | 6.67+/-0.88 | 1.33+/-0.08 |
| M-12-Creek Bank | 1262279+/-511.38 | 9.00+/-0.58 | 1.05+/-0.18 |
| M-0-Creek Bank | 528778+/-57.36 | 8.00+/-0.58 | 1.12+/-0.13 |
| **July** | | | |
| P-Creek Bank | 35447+/-15.34 | 7.68+/-1.45 | 1.64+/-0.14 |
| P-Edge of Vegetation | 100811+/-30.61 | 6.00+/-0.58 | 1.06+/-0.22 |
| S-Creek Bank | 46928+/-28.83 | 7+/-1.53 | 1.50+/-0.18 |
| S-Edge of Vegetation | 122097+/-31.54 | 4.67+/-0.67 | 0.71+/-0.12 |
| M-12-Creek Bank | 1660916+/-622.00 | 10.00+/-0.58 | 0.94+/-0.17 |
| M-12-Edge of Vegetation | 554756+/-49.11 | 8.33+/-0.33 | 1.13+/-0.02 |
| M-0-Creek Bank | 578220+/-106.29 | 7.33+/-0.88 | 0.79+/-0.09 |
| M-0-Edge of Vegetation | 611991+/-158.60 | 5.67+/-0.33 | 0.94+/-0.09 |
| **August** | | | |
| P-Creek Bank | 100309+/-30.59 | 10.33+/-0.88 | 1.77+/-0.03 |
| P-Edge of Vegetation | 125113+/-36.43 | 9.67+/-0.88 | 1.37+/-0.08 |
| S-Creek Bank | 84889+/-41.38 | 8.67+/-0.33 | 1.65+/-0.14 |
| S-Edge of Vegetation | 240506+/-80.48 | 5.33+/-0.88 | 1.06+/-0.07 |
| M-12-Creek Bank | 2139414+/-720.27 | 10.67+/-0.33 | 0.66+/-0.08 |
| M-12-Edge of Vegetation | 1670721+/-608.09 | 7.67+/-0.67 | 0.80+/-0.10 |
| M-0-Creek Bank | 2508134+/-574.03 | 7.00+/-0.00 | 0.37+/-0.06 |
| M-0-Edge of Vegetation | 343329+/-71.01 | 7.00+/-1.00 | 1.33+/-0.10 |
| **September** | | | |

| | | | |
|---|---|---|---|
| P-Creek Bank | 38799+/-20.38 | 6.67+/-1.45 | 1.49+/-0.19 |
| P-Edge of Vegetation | 85225+/-6.57 | 6.33+/-0.33 | 1.33+/-0.14 |
| S-Creek Bank | 91342+/-14.47 | 7.67+/-0.88 | 1.30+/-0.06 |
| S-Edge of Vegetation | 109778+/-3.46 | 6.00+/-0.58 | 1.11+/-0.14 |
| M-12-Creek Bank | 1791393+/-446.71 | 9.67+/-0.67 | 0.63+/-0.13 |
| M-12-Edge of Vegetation | 3470745+/-506.24 | 6.67+/-0.67 | 0.33+/-0.08 |
| M-0-Creek Bank | 1652536+/-119.98 | 5.67+/-0.33 | 0.38+/-0.02 |
| M-0-Edge of Vegetation | 128801+/-4.91 | 6.67+/-0.33 | 1.31+/-0.10 |

**Table 4. P values and degrees of freedom for abundance, taxa richness, and Shannon-Wiener Diversity Index at Sawmill Creek—*P. australis* (P), Sawmill Creek—*S. alterniflora* (S), 12-year-old mitigation site—Mill Creek (M-12), and new mitigation site—Mill Creek (M-0).**

| Site | Abundance | Taxa Richness | Shannon-Wiener Diversity Index |
|---|---|---|---|
| **June** | | | |
| P-Creek Bank<br>S-Creek Bank | P=0.0962/df=1 | P=0.0739/df=1 | P=0.0347/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.1487/df=1 | P=0.2879/df=1 | P=0.7384/df=1 |
| **July** | | | |
| P-Creek Bank<br>P-Edge of Vegetation | P=0.0845/df=1 | P=0.3465/df=1 | P=0.1002/df=1 |
| S-Creek Bank<br>S-Edge of Vegetation | P=0.1061/df=1 | P=0.2341/df=1 | P=0.0218/df=1 |
| P-Creek Bank<br>S-Creek Bank | P=0.8046/df=1 | P=0.7676/df=1 | P=0.5818/df=1 |
| P-Edge of Vegetation<br>S-Edge of Vegetation | P=0.5908/df=1 | P=0.2051/df=1 | P=0.2410/df=1 |
| M-12-Creek Bank<br>M-12-Edge of Vegetation | P=0.0357/df=1 | P=0.0668/df=1 | P=0.1653/df=1 |
| M-0-Creek Bank<br>M-0-Edge of Vegetation | P=0.8881/df=1 | P=0.1518/df=1 | P=0.2609/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.0502/df=1 | P=0.0647/df=1 | 0.4636/df=1 |
| M-12-Edge of Vegetation<br>M-0-Edge of Vegetation | P=0.8001/df=1 | P=0.0048/df=1 | P=0.0333/df=1 |
| **August** | | | |
| P-Creek Bank<br>P-Edge of Vegetation | P=0.5634/df=1 | P=0.6213/df=1 | P=0.0081/df=1 |
| S-Creek Bank<br>S-Edge of Vegetation | P=0.0962/df=1 | P=0.0241/df=1 | P=0.0189/df=1 |
| P-Creek Bank<br>S-Creek Bank | P=0.6425/df=1 | P=0.1518/df=1 | P=0.4453/df=1 |
| P-Edge of Vegetation | P=0.1959/df=1 | P=0.0255/df=1 | P=0.0371/df=1 |

| | | | |
|---|---|---|---|
| S-Edge of Vegetation | | | |
| M-12-Creek Bank<br>M-12-Edge of Vegetation | P=0.6105/df=1 | P=0.0158/df=1 | P=0.3445/df=1 |
| M-0-Creek Bank<br>M-0-Edge of Vegetation | P=0.0024/df=1 | P=1.0000/df=1 | P=0.0012/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.6110/df=1 | P=0.0004/df=1 | P=0.0464/df=1 |
| M-12-Edge of Vegetation<br>M-0-Edge of Vegetation | P=0.0133/df=1 | P=0.6087/df=1 | P=0.0225/df=1 |
| **September** | | | |
| P-Creek Bank<br>P-Edge of Vegetation | P=0.1226/df=1 | P=0.8340/df=1 | P=0.5500/df=1 |
| S-Creek Bank<br>S-Edge of Vegetation | P=0.2151/df=1 | P=0.1890/df=1 | P=0.2565/df=1 |
| P-Creek Bank<br>S-Creek Bank | P=0.1149/df=1 | P=0.5879/df=1 | P=0.4242/df=1 |
| P-Edge of Vegetation<br>S-Edge of Vegetation | P=0.0212/df=1 | P=0.6433/df=1 | P=0.3270/df=1 |
| M-12-Creek Bank<br>M-12-Edge of Vegetation | P=0.0601/df=1 | P=0.0335/df=1 | P=0.1173/df=1 |
| M-0-Creek Bank<br>M-0-Edge of Vegetation | P<0.0001/df=1 | P=0.1012/df=1 | P=0.0007/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.8974/df=1 | P=0.0058/df=1 | P=0.1371/df=1 |
| M-12-Edge of Vegetation<br>M-0-Edge of Vegetation | P<0.0001/df=1 | P=1.0000/df=1 | P=0.0014/df=1 |

**Table 5. Taxa present at (+) or absent from (–) Sawmill Creek—*P. australis* (P), Sawmill Creek—*S. alterniflora* (S), 12-year-old mitigation site—Mill Creek (M-12), and new mitigation site—Mill Creek (M-0); sampled from June to September 1999.**

| **Site** | **Abundance** | **Taxa Richness** | **Shannon-Wiener Diversity Index** |
|---|---|---|---|
| **June** | | | |
| P-Creek Bank<br>S-Creek Bank | P=0.0962/df=1 | P=0.0739/df=1 | P=0.0347/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.1487/df=1 | P=0.2879/df=1 | P=0.7384/df=1 |
| **July** | | | |
| P-Creek Bank<br>P-Edge of Vegetation | P=0.0845/df=1 | P=0.3465/df=1 | P=0.1002/df=1 |
| S-Creek Bank<br>S-Edge of Vegetation | P=0.1061/df=1 | P=0.2341/df=1 | P=0.0218/df=1 |
| P-Creek Bank<br>S-Creek Bank | P=0.8046/df=1 | P=0.7676/df=1 | P=0.5818/df=1 |
| P-Edge of Vegetation<br>S-Edge of Vegetation | P=0.5908/df=1 | P=0.2051/df=1 | P=0.2410/df=1 |
| M-12-Creek Bank<br>M-12-Edge of Vegetation | P=0.0357/df=1 | P=0.0668/df=1 | P=0.1653/df=1 |
| M-0-Creek Bank<br>M-0-Edge of Vegetation | P=0.8881/df=1 | P=0.1518/df=1 | P=0.2609/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.0502/df=1 | P=0.0647/df=1 | 0.4636/df=1 |
| M-12-Edge of Vegetation<br>M-0-Edge of Vegetation | P=0.8001/df=1 | P=0.0048/df=1 | P=0.0333/df=1 |
| **August** | | | |
| P-Creek Bank | P=0.5634/df=1 | P=0.6213/df=1 | P=0.0081/df=1 |

| | | | |
|---|---|---|---|
| P-Edge of Vegetation | | | |
| S-Creek Bank<br>S-Edge of Vegetation | P=0.0962/df=1 | P=0.0241/df=1 | P=0.0189/df=1 |
| P-Creek Bank<br>S-Creek Bank | P=0.6425/df=1 | P=0.1518/df=1 | P=0.4453/df=1 |
| P-Edge of Vegetation<br>S-Edge of Vegetation | P=0.1959/df=1 | P=0.0255/df=1 | P=0.0371/df=1 |
| M-12-Creek Bank<br>M-12-Edge of Vegetation | P=0.6105/df=1 | P=0.0158/df=1 | P=0.3445/df=1 |
| M-0-Creek Bank<br>M-0-Edge of Vegetation | P=0.0024/df=1 | P=1.0000/df=1 | P=0.0012/df=1 |
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.6110/df=1 | P=0.0004/df=1 | P=0.0464/df=1 |
| M-12-Edge of Vegetation<br>M-0-Edge of Vegetation | P=0.0133/df=1 | P=0.6087/df=1 | P=0.0225/df=1 |
| **September** | | | |
| P-Creek Bank<br>P-Edge of Vegetation | P=0.1226/df=1 | P=0.8340/df=1 | P=0.5500/df=1 |
| S-Creek Bank<br>S-Edge of Vegetation | P=0.2151/df=1 | P=0.1890/df=1 | P=0.2565/df=1 |
| P-Creek Bank<br>S-Creek Bank | P=0.1149/df=1 | P=0.5879/df=1 | P=0.4242/df=1 |
| P-Edge of Vegetation<br>S-Edge of Vegetation | P=0.0212/df=1 | P=0.6433/df=1 | P=0.3270/df=1 |
| M-12-Creek Bank<br>M-12-Edge of Vegetation | P=0.0601/df=1 | P=0.0335/df=1 | P=0.1173/df=1 |
| M-0-Creek Bank<br>M-0-Edge of Vegetation | P<0.0001/df=1 | P=0.1012/df=1 | P=0.0007/df=1 |

| | | | |
|---|---|---|---|
| M-12-Creek Bank<br>M-0-Creek Bank | P=0.8974/df=1 | P=0.0058/df=1 | P=0.1371/df=1 |
| M-12-Edge of Vegetation<br>M-0-Edge of Vegetation | P<0.0001/df=1 | P=1.0000/df=1 | P=0.0014/df=1 |

Note: Sawmill Creek and Mill Creek sites should not be compared, even though they are both represented in this table.

**Table 6. Totals and percent totals of the common and uncommon species.**

| **Sawmill Creek—*Phragmites australis*** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Common Species** | *Cyanthura polita* | Ceratopogonidae | Nereidae | Spionidae | *Hobsonia florida* | *Streblospio benedicti* | Copepoda | *Manayunkia aestuarina* | Nematoda | Oligochaeta |
| Total | 23 | 26 | 57 | 75 | 121 | 138 | 296 | 386 | 526 | 559 |
| % of Total | 1.00 | 1.13 | 2.48 | 3.27 | 5.27 | 6.02 | 12.90 | 16.83 | 22.93 | 24.37 |
| **Uncommon Species** | *Macoma bathica* | *Anurida martina* | Acarina | *Edotea triloba* | Chironomidae | Foraminifera | Turbellaria | Insect Larvae | | |
| Total | 3 | 4 | 5 | 5 | 6 | 14 | 19 | 22 | | |
| % of Total | 0.13 | 0.17 | 0.22 | 0.22 | 0.26 | 0.61 | 0.83 | 0.96 | | |
| **Sawmill Creek—*Spartina alterniflora*** | | | | | | | | | | |
| **Common Species** | Spionidae | *Hobsonia florida* | Nereidae | Copepoda | *Streblospio benedicti* | Oligochaeta | Nematoda | *Manayunkia aestuarina* | | |
| Total | 30 | 41 | 62 | 126 | 186 | 542 | 873 | 940 | | |
| % of Total | 1.05 | 1.44 | 2.18 | 4.43 | 6.54 | 19.04 | 30.67 | 33.03 | | |
| **Uncommon Species** | Gammarid | Foraminifera | Ceratopogonidae | *Cyanthura polita* | Turbellaria | | | | | |
| Total | 3 | 7 | 9 | 10 | 11 | | | | | |
| % of Total | 0.11 | 0.25 | 0.32 | 0.35 | 0.39 | | | | | |
| **New Mitigation Site —Mill Creek** | | | | | | | | | | |
| **Common Species** | Gastropoda | Ostracoda | Oligochaeta | Nematoda | | | | | | |
| Total | 459 | 1540 | 2859 | 17571 | | | | | | |
| %of Total | 2.02 | 6.77 | 12.57 | 77.27 | | | | | | |
| **Uncommon Species** | *Hobsonia florida* | Chrionomidae | Copepoda | | | | | | | |
| Total | 31 | 104 | 147 | | | | | | | |
| %of Total | 0.14 | 0.46 | 0.65 | | | | | | | |
| **12-yr-old Mitigation Site —Mill Creek** | | | | | | | | | | |
| **Common Species** | *Manayunkia aestuarina* | Copepoda | Oligochaeta | Nematoda | | | | | | |
| Total | 826 | 1260 | 6168 | 35853 | | | | | | |
| %of Total | 1.84 | 2.80 | 13.73 | 79.78 | | | | | | |
| **Uncommon Species** | Gastropoda | Ostracoda | | | | | | | | |
| Total | 273 | 362 | | | | | | | | |
| %of Total | 0.61 | 0.81 | | | | | | | | |

# The Changing Flora of the New York Metropolitan Region*

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## Abstract

We statistically analyzed 100 years of herbarium specimen data for woody plants in the New York metropolitan region in order to measure the floristic changes of this area. Change index values were computed for 224 of the region's 556 woody species to provide a specific measure of whether these species are expanding, contracting, or stable. The results show that, in general, nonnative invasive species are spreading rapidly in the region, while native species are in slight decline.

**Keywords:** *Chimaphila*, ecological change; Ericaceae; herbarium; invasive plants; *Lonicera*; New York City, urban flora

## Introduction

Plant species differ in their ability to adapt to environmental changes brought on by urban development and spread. Yet there are few studies that attempt to quantify the differences in adaptability among species (but see, for example, Dickson et al., 2000). In this study, we use current and historical data on woody plants in the New York metropolitan region to develop a change index measuring the relative degree to which species have expanded or contracted their ranges over the past century. The findings help us gain a better understanding of exactly how the flora of this urban region is changing and should prove useful to those attempting to improve and restore the ecosystems of the region.

It is difficult to quantify changes in the flora of the New York metropolitan region because the region, like other urban areas in the United States, has not been subjected to any long-term plant studies using standard sampling methods. In our study, we used herbarium specimen data from about a dozen herbaria in the northeastern United States. Botanists do not use a standard sampling method when collecting herbarium specimens: Some collect every plant they see, while others collect only the plants they are studying or those that are of particular interest at a site. But although there are a variety of sampling strategies, the strategies themselves have not changed significantly over the past century, and the data should be adequate for carrying out a comparison of the relative changes in the ranges of species.

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Although our technique only analyzes the change in range of a species, it has been shown that there is a relationship between range and abundance of species (Hanski, Kouki & Halkka, 1993; He, Gaston & Wu, 2002). Therefore, an expanding range for a species is a good indication that the species may be increasing in abundance. Likewise, a range contraction is an indicator that a species may be declining in abundance.

## Methods

This study is comparable to a study done for plants in Great Britain. We have predominantly used techniques developed by Telfer, Preston, and Rothery (2002), with a few modifications, spelled out in detail here.

The distributional data comes from the New York Metropolitan Flora (NYMF) project database (Moore, Steward, Clemants, Glenn & Ma, 2002; and see http://www.bbg.org/sci/nymf/). This database currently has nearly 250,000 records of plant occurrences from the New York metropolitan region. Each record is geo-coded to five-kilometer-square cells in a grid, with 964 cells total. We will call these cells "blocks." (The names used in this study are those adopted by the NYMF project; see Moore et al., 2002.)

In this study, we used the woody-species data from the NYMF database. The woody-plant data set is the most complete one in the database and has over 145,000 records, representing 556 species. In our analysis, we only used records of woody species based upon herbarium specimens collected between 1901 and 2000. Once we narrowed the data to meet this criterion and eliminated duplicate records, there were 24,795 records remaining for this study. These records were made relatively evenly over the first half of the 20th century, but for the second half of the century, the bulk of the data is from the last decade (the 1990s), when the NYMF project began actively collecting (Figure 1).

The data were partitioned into two cohorts (time periods): the early cohort, containing data from 1901 to 1950, and the later cohort, containing data from 1951 to 2000. Following Telfer, Preston, and Rothery (2002), we only included blocks for which there were occurrences of a species in both cohorts. This reduced the number of blocks used in the analysis to 647. These 647 blocks are distributed throughout the New York metropolitan region (Figure 2). The Telfer, Preston, and Rothery study excluded species with fewer than five occurrences in the early cohort. In our study, we modified the procedure by excluding species with fewer than five occurrences in either the early or late cohort. This reduced the number of species in our study to 224.

The statistical methods for developing the change index are outlined in Telfer, Preston, and Rothery (2002). All statistics were calculated using Systat 10.2 statistical software (SPSS, 2000).

## Results and Discussion

Table 1 lists the 224 species studied in this analysis, the raw sampling block counts for each cohort, species provenance (native or introduced), and the change index. Please note that the raw counts for some species show an increase over time, while their change indices show a decrease. This is because there are many more records in the later period (from the 1990s). The statistic essentially corrects for this overabundance of data. This means that a species showing no change in distribution will have a larger raw count in the later period than the earlier, and that some species may show a decrease in distribution while showing an increase in the raw count.

The first, unweighted least-squares regression equation was $y = -1.05 + 0.66x$, with $r^2 = 0.444$. Following two iterations of the weighing procedure, we arrived at a weighted regression equation of $y = -1.00 + 0.68x$, with $r^2 = 0.467$. We believe that the relatively low $r^2$ is the result of two things. First, unlike in Telfer, Preston, and Rothery (2002), our data were not collected following a uniform procedure. Therefore, we suspect that there is greater statistical error in the data. Second, we believe we are studying a much more rapidly changing flora (an urban flora) than the one in the studies used by Telfer, Preston, and Rothery (a country-wide flora). Therefore, we would expect larger change indices in general.

Figure 3 shows the distribution of change indices in relation to the provenance of the plant species. Because the data for natives are right-skewed, we used a Mann-Whitney U test to determine if the native and nonnative (introduced) species data are significantly different. The Mann-Whitney test statistic was 5054, which is significant ($p = 0.014$). This indicates that the nonnative species are increasing relative to the native species. In general, native species are showing slight decline, and introduced species are showing much greater expansion of their ranges, with only a few species showing any decline.

The change index in this study is valuable because it provides species-specific information about what is changing in the flora. For instance, nearly all the members of the heath family (Ericaceae) in the region are showing contraction of their ranges. There are probably many reasons why these species appear sensitive to urbanization, but three stand out: 1) most heath family species are acidophilic (Kron & Chase, 1993), and urban soils are generally more basic (Craul, 1992; Scheyer & Hipple, 2005); 2) many Ericaceae species are hydrophytes, and much wetland habitat has been lost over the past century (e.g., New Jersey lost an estimated 39% of its wetlands between 1870 and 1970, with half that loss occurring between 1950 and 1970; see New Jersey Sustainable State Institute, 2004); 3) the overabundance of white-tailed deer (*Odocoileus virginianus*) in suburban regions may impact some species through overgrazing (Department of Environmental Protection, Division of Fish, Game and Wildlife, 1999), though we expect this impact would be broad across many taxa .

The results show that several congeneric species have very different change indices. For example, *Celastrus scandens*, the native American bittersweet, has a change index of –1.15, while *Celastrus orbiculata*, the nonnative Oriental bittersweet, has a change index of +3.24. This wide disparity—indicative of a dramatic decline for the American bittersweet and a dramatic spread by the Oriental bittersweet—reinforces the results of a previously published account of these two species (Steward, Clemants & Moore, 2003).

Nonnative honeysuckles are significantly increasing, while native species are undergoing significant decline. The native *Lonicera dioica* and *L. sempervirens* have change indices of –2.87 and –1.93, respectively, and the nonnative *L. japonica* and *L. morrowii* have change indices of +1.60 and +1.73, respectively (see Figures 4–7). (In the case of *L. japonica* and *L. sempervirens*, the nonnative's growth architecture may be giving it a competitive advantage over its native congener and allowing it to increase its range; see Schweitzer & Larson, 1990; Larson, 2000).

Another nonnative species, *L. maackii*, not included in this study because of its more recent date of introduction (and thus lack of any pre-1950 records), is also rapidly spreading in the region (Figure 8).

Other native-nonnative congeneric species groups also reflect this pattern, such as the following (change index in parentheses): nonnative *Clematis terniflora* (+1.33), native *C. virginiana* (–0.32); nonnative *Morus alba* (+2.41), native *M. rubra* (–1.71); nonnative *Ribes rubrum* (+0.28), native *R. americanum* (–0.41), native *R. hirtellum* (–1.92), and native *R. rotundifolium* (–0.54).

A striking pattern is observed for the New York metropolitan region's two native *Chimaphila* species (which are not being impacted by nonnative congeners), with *C. umbellata* having a change index of –2.51 and *C. maculata* having a change index of –0.29 (Figures 9 and 10). While there have not been any studies aimed at better understanding why *C. umbellata* is declining at a greater rate than *C. maculata*, field botanists have hypothesized that *C. umbellata* may be more significantly affected by deer browsing than *C. maculata*, perhaps as a result of differences in leaf chemistry between the two species (Lamont & Young, 2004). Cowan (1945) reported that *C. umbellata* was casually eaten by deer.

## Conclusion

Without question, the flora of the New York metropolitan region is rapidly changing. Most notably, nonnative invasive species are rapidly spreading in the area, while native species are generally in decline. Monitoring programs such as the NYMF project provide a mechanism by which these changes can be quantitatively measured. They may, in the future, be used to identify potentially invasive species before these species spread throughout the range. Also, these programs provide baseline data that future generations can use in comparative analysis to track floristic change.

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## Glossary

**Acidophilic:** Pertaining to plants that thrive in acid soil.

**Basic:** Alkaline

**Change Index:** A statistical indication of changes in the distribution of a species. A positive change index indicates that a species is expanding its range, while a negative change indicates that a species is contracting its range.

**Congeneric, congener:** Belonging to the same genus.

**Geo-code:** A computerized process that uses coordinates (in our case, cells) to uniquely identify a geographic location from a description.

**Hydrophyte:** An aquatic plant; one that grows in water or needs a waterlogged habitat.

**Least-squares regression equation:** A statistical method for a simple linear equation to real data points.

**Mann-Whitney U test:** A non-parametric test used to compare two independent groups of sampled data.

**Figure 1. Number of unique specimens of woody species collected over the past century.**

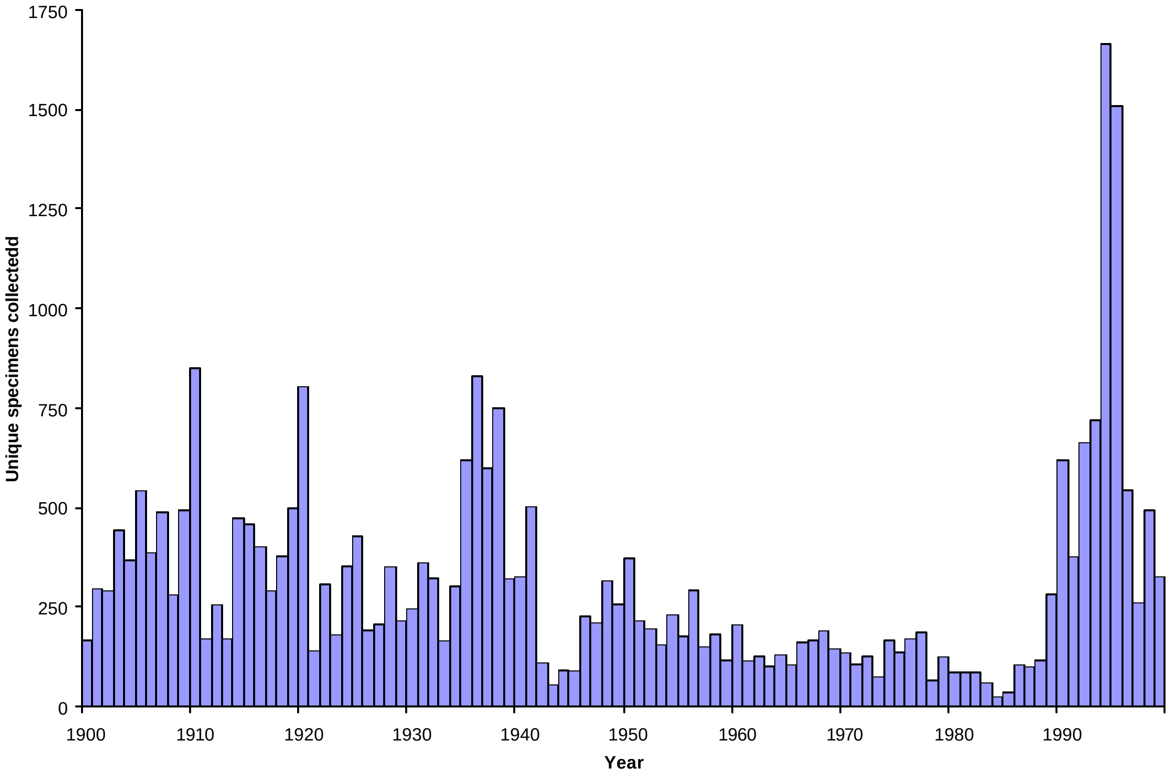


**Figure 2. Distribution of blocks used in this study.**

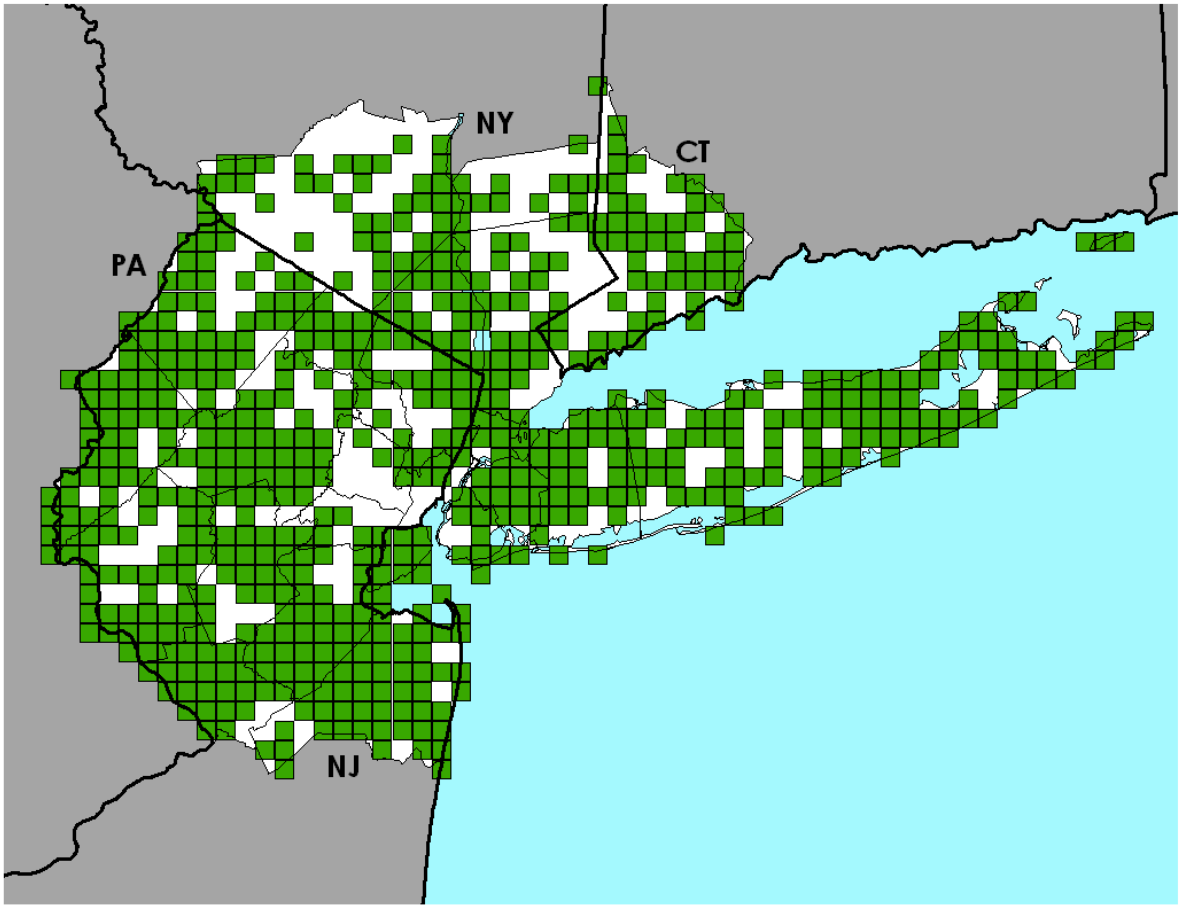


**Figure 3. A dual histogram of the change indices for introduced (nonnative) and native species. These graphs show the distribution of change index values for the 226 species studied.**

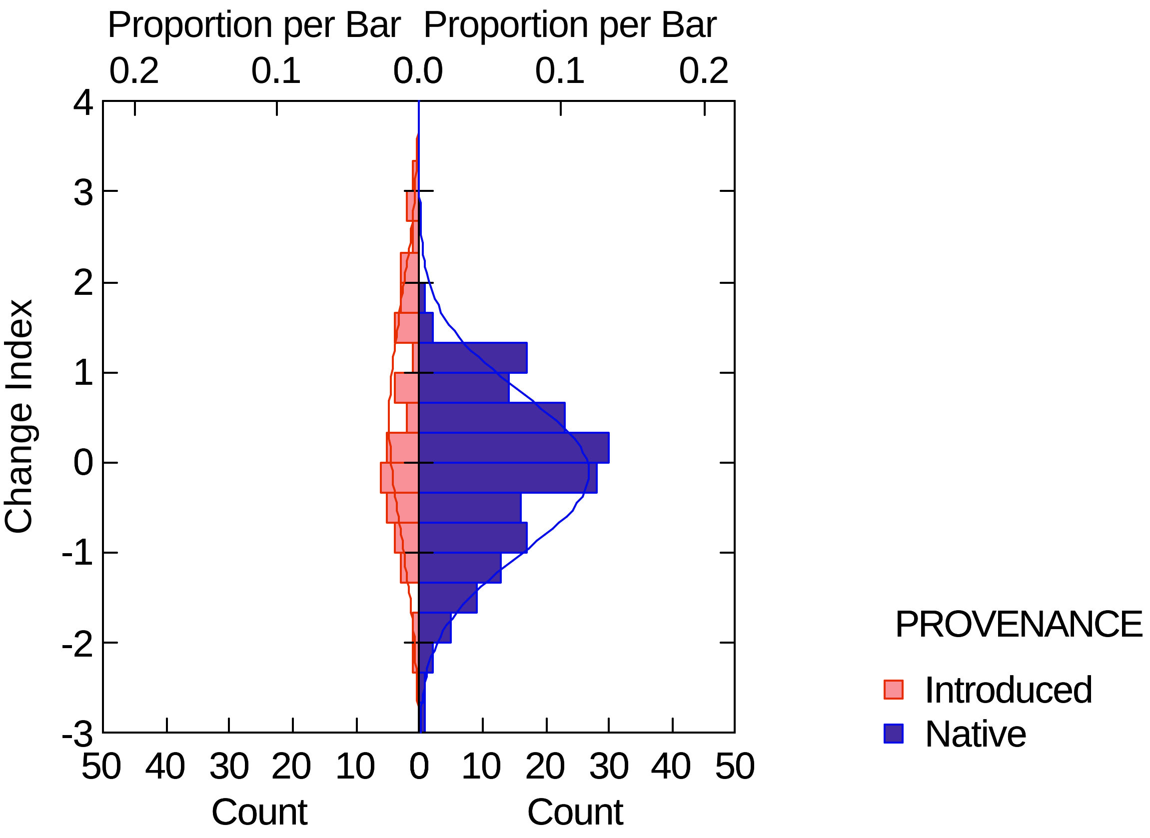
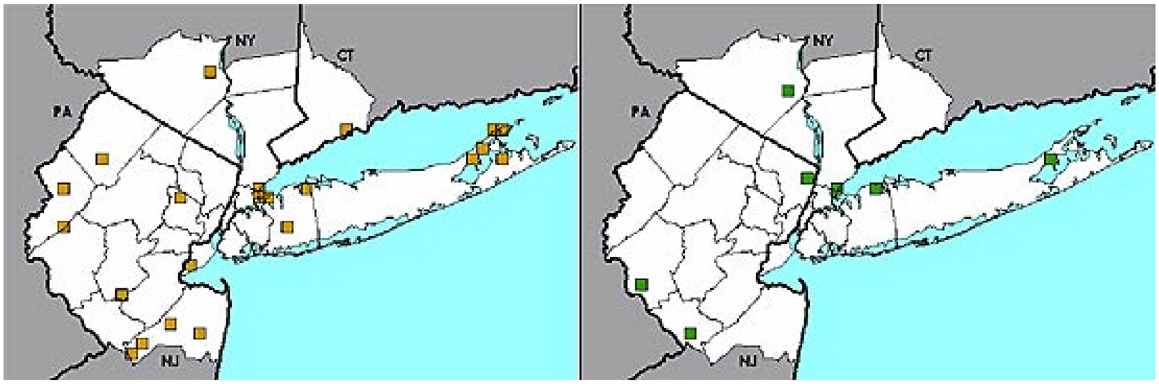


**Figure 4. Range map of *Lonicera sempervirens* for the New York metropolitan area. (Native, Change Index = –1.93)**



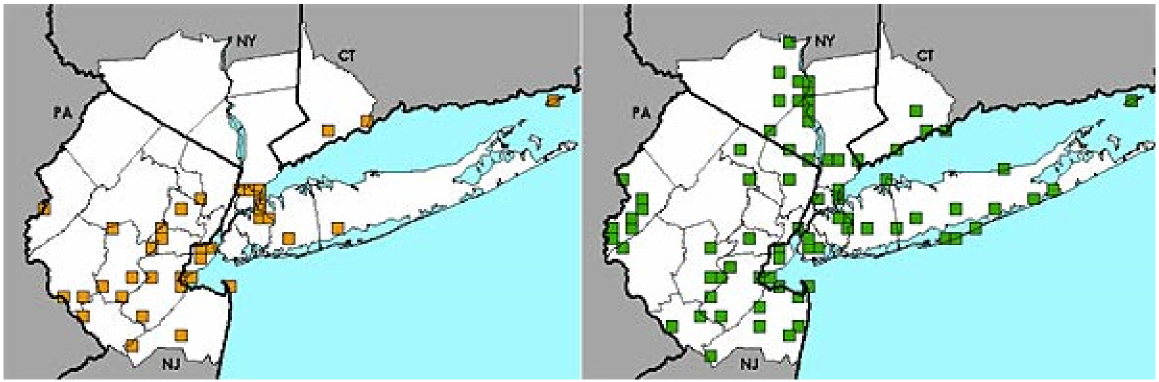
4a 4b

(Native, Change Index = –1.93)

4a. Specimens collected between 1901 and 1950

4b. Specimens collected between 1951 and 2000

**Figure 5. Range map of *Lonicera japonica* for the New York metropolitan area. (Introduced, Change Index = +1.60)**



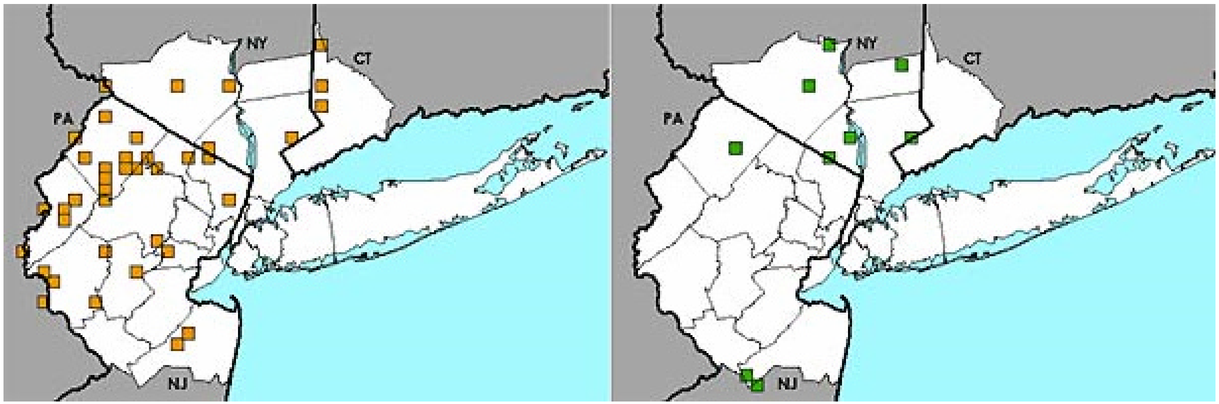
5a 5b

(Introduced, Change Index = +1.60)

5a. Specimens collected between 1901 and 1950

5b. Specimens collected between 1951 and 2000

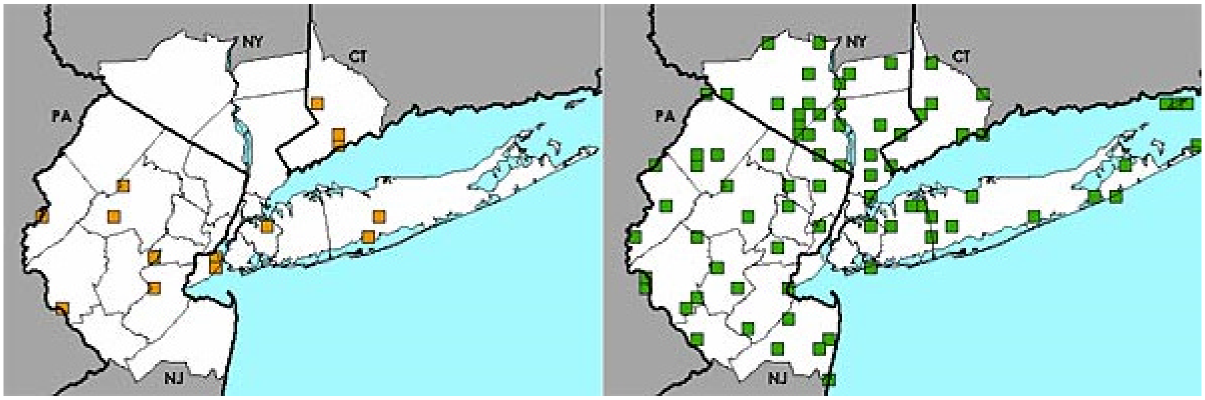
**Figure 6. Range map of *Lonicera dioica* for the New York metropolitan area.**



6a 6b

(Native, Change Index = –2.87)
6a. Specimens collected between 1901 and 1950
6b. Specimens collected between 1951 and 2000

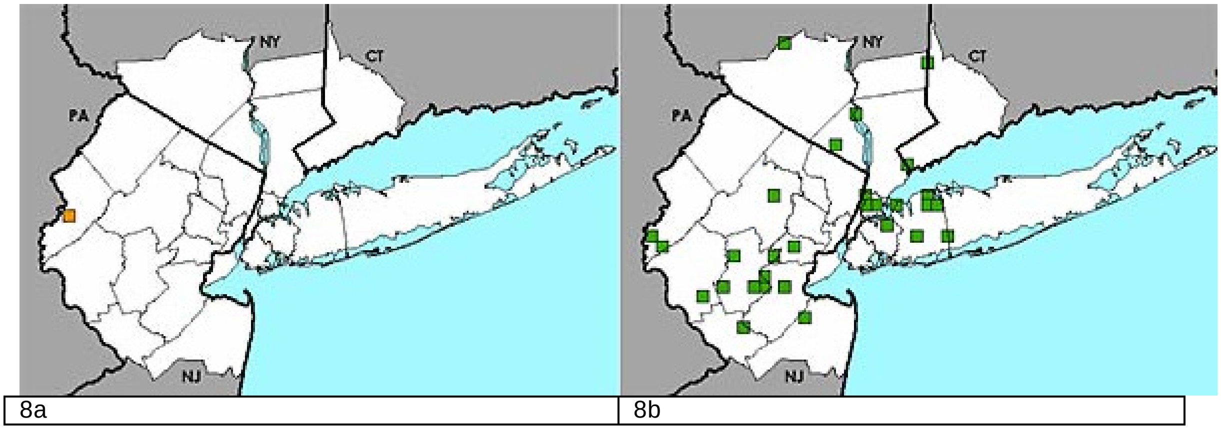
**Figure 7. Range map of *Lonicera morrowii* for the New York metropolitan area.**



7a 7b

(Native, Change Index = –2.73)
7a. Specimens collected between 1901 and 1950
7b. Specimens collected between 1951 and 2000

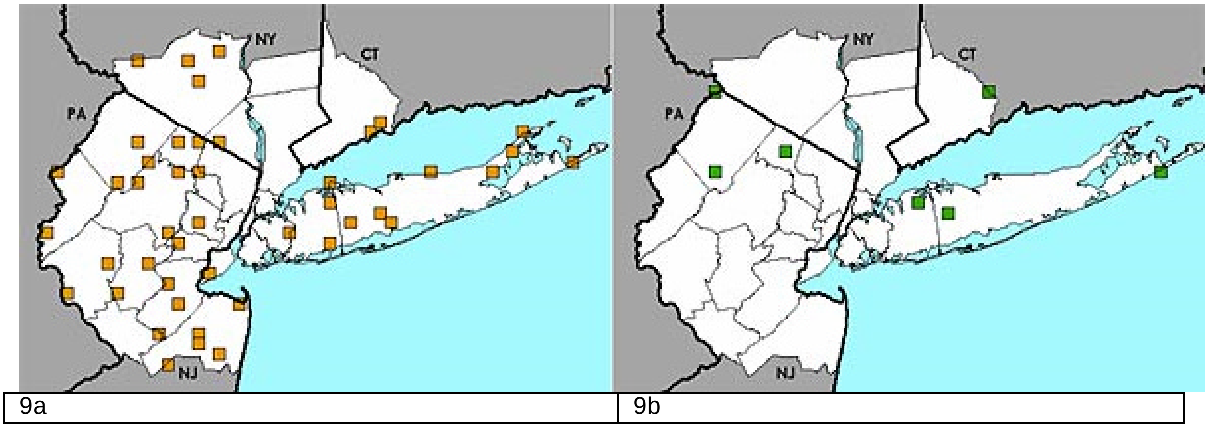
**Figure 8. Range map of *Lonicera maackiii* for the New York metropolitan area.**



8a
8b

(Introduced, no change index, too few collections in early period)
8a. Specimens collected between 1901 and 1950
8b. Specimens collected between 1951 and 2000

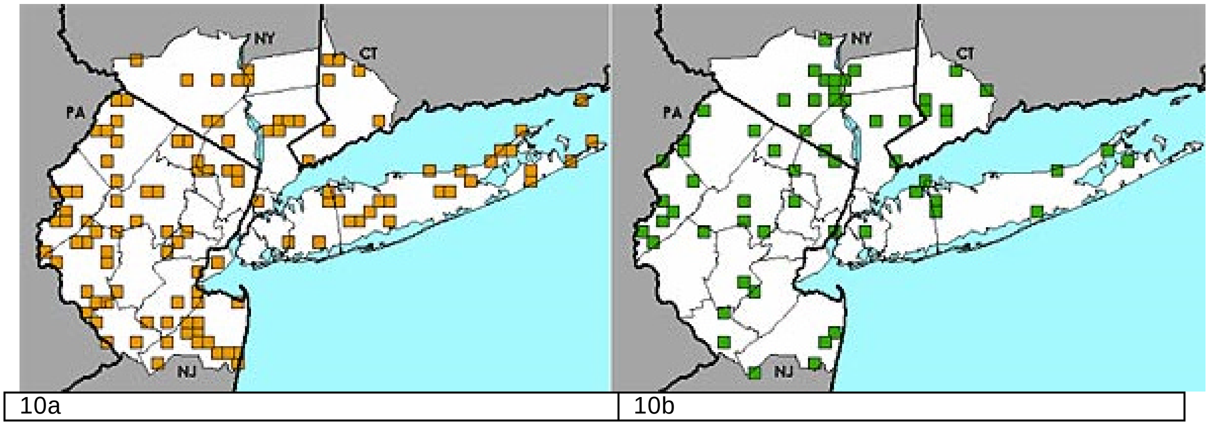
**Figure 9. Range map of *Chimaphila umbellata* for the New York metropolitan area.**



9a
9b

(Native, Change Index = –2.51)
9a. Specimens collected between 1901 and 1950
9b. Specimens collected between 1951 and 2000

**Figure 10. Range map of *Chimaphila maculata* for the New York metropolitan area.**



| 10a | 10b |
|---|---|

(Native, Change Index = –0.29)
10a. Specimens collected between 1901 and 1950
10b. Specimens collected between 1951 and 2000

**Table 1. The change index for each species in the study along with the raw data and the provenance of each species. (Names follow Moore et al., 2002.)**

| **Species name** | **Provenance** | **1901–1950 raw count** | **1951–2000 raw count** | **Change Index** |
|---|---|---|---|---|
| *Acer negundo* | Native | 22 | 65 | 1.86 |
| *Acer pensylvanicum* | Native | 18 | 23 | 0.20 |
| *Acer platanoides* | Introduced | 22 | 58 | 1.64 |
| *Acer pseudoplatanus* | Introduced | 13 | 23 | 0.57 |
| *Acer rubrum* | Native | 93 | 117 | 1.26 |
| *Acer saccharinum* | Native | 22 | 39 | 0.91 |
| *Acer saccharum* | Native | 45 | 69 | 1.12 |
| *Acer spicatum* | Native | 26 | 11 | –1.48 |
| *Aesculus hippocastanum* | Introduced | 6 | 8 | –0.37 |
| *Ailanthus altissima* | Introduced | 16 | 54 | 1.88 |
| *Akebia quinata* | Introduced | 6 | 6 | –0.84 |
| *Alnus incana* | Native | 17 | 20 | 0.03 |
| *Alnus serrulata* | Native | 85 | 67 | 0.26 |
| *Amelanchier arborea* | Native | 29 | 43 | 0.76 |
| *Amelanchier canadensis* | Native | 47 | 90 | 1.59 |
| *Amelanchier stolonifera* | Native | 15 | 16 | –0.22 |
| *Amorpha fruticosa* | Native | 22 | 37 | 0.81 |
| *Ampelopsis brevipedunculata* | Introduced | 8 | 40 | 2.12 |
| *Aralia spinosa* | Introduced | 6 | 34 | 2.14 |
| *Arctostaphylos uva-ursi* | Native | 43 | 13 | –1.81 |
| *Aronia arbutifolia* | Native | 71 | 59 | 0.25 |
| *Aronia melanocarpa* | Native | 39 | 21 | –0.86 |
| *Baccharis halimifolia* | Native | 37 | 39 | 0.30 |
| *Berberis thunbergii* | Introduced | 25 | 65 | 1.71 |
| *Berberis vulgaris* | Introduced | 17 | 12 | –0.84 |
| *Betula alleghaniensis* | Native | 34 | 21 | –0.69 |
| *Betula lenta* | Native | 69 | 64 | 0.44 |
| *Betula nigra* | Native | 30 | 21 | –0.55 |
| *Betula papyrifera* | Native | 12 | 7 | –1.35 |
| *Betula populifolia* | Native | 82 | 74 | 0.50 |
| *Broussonetia papyrifera* | Introduced | 15 | 10 | –1.01 |

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|---|---|---|---|---|
| *Campsis radicans* | Introduced | 10 | 15 | 0.13 |
| *Carpinus caroliniana* | Native | 53 | 66 | 0.83 |
| *Carya cordiformis* | Native | 20 | 40 | 1.06 |
| *Carya glabra* | Native | 42 | 46 | 0.44 |
| *Carya ovalis* | Native | 15 | 12 | –0.70 |
| *Carya ovata* | Native | 28 | 43 | 0.80 |
| *Carya tomentosa* | Native | 50 | 51 | 0.42 |
| *Castanea dentata* | Native | 63 | 50 | 0.10 |
| *Catalpa bignonioides* | Introduced | 10 | 24 | 0.94 |
| *Ceanothus americanus* | Native | 61 | 25 | –1.10 |
| *Celastrus orbiculata* | Introduced | 8 | 71 | 3.24 |
| *Celastrus scandens* | Native | 81 | 30 | –1.15 |
| *Celtis occidentalis* | Native | 68 | 56 | 0.21 |
| *Cephalanthus occidentalis* | Native | 53 | 63 | 0.74 |
| *Chamaecyparis thyoides* | Native | 27 | 20 | –0.51 |
| *Chamaedaphne calyculata* | Native | 49 | 23 | –0.98 |
| *Chimaphila maculata* | Native | 107 | 59 | –0.29 |
| *Chimaphila umbellata* | Native | 39 | 8 | –2.51 |
| *Clematis terniflora* | Introduced | 8 | 26 | 1.33 |
| *Clematis virginiana* | Native | 36 | 27 | –0.32 |
| *Clethra alnifolia* | Native | 101 | 63 | –0.08 |
| *Comptonia peregrina* | Native | 63 | 46 | –0.05 |
| *Cornus alternifolia* | Native | 41 | 34 | –0.07 |
| *Cornus amomum* | Native | 75 | 75 | 0.64 |
| *Cornus florida* | Native | 87 | 83 | 0.65 |
| *Cornus foemina* | Native | 77 | 64 | 0.30 |
| *Cornus rugosa* | Native | 31 | 18 | –0.85 |
| *Cornus sericea* | Native | 12 | 21 | 0.50 |
| *Corylus americana* | Native | 60 | 56 | 0.37 |
| *Corylus cornuta* | Native | 21 | 16 | –0.60 |
| *Crataegus crusgalli* | Native | 17 | 14 | –0.58 |
| *Crataegus pruinosa* | Native | 21 | 13 | –0.95 |
| *Diervilla lonicera* | Native | 38 | 19 | –1.00 |
| *Diospyros virginiana* | Native | 20 | 16 | –0.54 |

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|---|---|---|---|---|
| *Dirca palustris* | Native | 8 | 6 | –1.15 |
| *Elaeagnus umbellata* | Introduced | 12 | 53 | 2.18 |
| *Epigaea repens* | Native | 67 | 26 | –1.16 |
| *Euonymus europaea* | Introduced | 19 | 12 | –0.97 |
| *Fagus grandifolia* | Native | 42 | 71 | 1.26 |
| *Fraxinus americana* | Native | 50 | 63 | 0.82 |
| *Fraxinus nigra* | Native | 21 | 27 | 0.31 |
| *Fraxinus pennsylvanica* | Native | 45 | 46 | 0.36 |
| *Gaultheria procumbens* | Native | 41 | 24 | –0.69 |
| *Gaylussacia baccata* | Native | 102 | 65 | –0.04 |
| *Gaylussacia frondosa* | Native | 59 | 28 | –0.86 |
| *Hamamelis virginiana* | Native | 66 | 73 | 0.75 |
| *Hibiscus syriacus* | Introduced | 7 | 10 | –0.16 |
| *Hudsonia ericoides* | Native | 30 | 8 | –2.19 |
| *Hudsonia tomentosa* | Native | 60 | 26 | –1.01 |
| *Hydrangea arborescens* | Native | 16 | 8 | –1.45 |
| *Ilex glabra* | Native | 32 | 15 | –1.20 |
| *Ilex laevigata* | Native | 24 | 17 | –0.65 |
| *Ilex opaca* | Native | 16 | 26 | 0.55 |
| *Ilex verticillata* | Native | 78 | 69 | 0.43 |
| *Iva frutescens* | Native | 34 | 33 | 0.10 |
| *Juglans cinerea* | Native | 21 | 23 | 0.03 |
| *Juglans nigra* | Native | 21 | 47 | 1.30 |
| *Juniperus communis* | Native | 19 | 10 | –1.28 |
| *Juniperus virginiana* | Native | 74 | 57 | 0.14 |
| *Kalmia angustifolia* | Native | 64 | 35 | –0.57 |
| *Kalmia latifolia* | Native | 67 | 49 | –0.02 |
| *Larix laricina* | Native | 14 | 11 | –0.77 |
| *Leucothoe racemosa* | Native | 76 | 39 | –0.59 |
| *Ligustrum vulgare* | Introduced | 13 | 14 | –0.28 |
| *Lindera benzoin* | Native | 73 | 97 | 1.18 |
| *Liquidambar styraciflua* | Native | 42 | 35 | –0.05 |
| *Liriodendron tulipifera* | Native | 32 | 61 | 1.29 |
| *Lonicera dioica* | Native | 35 | 6 | –2.87 |

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|---|---|---|---|---|
| *Lonicera japonica* | Introduced | 33 | 73 | 1.60 |
| *Lonicera morrowii* | Introduced | 14 | 77 | 2.73 |
| *Lonicera sempervirens* | Native | 20 | 7 | –1.93 |
| *Lycium barbarum* | Introduced | 13 | 10 | –0.85 |
| *Lyonia ligustrina* | Native | 104 | 54 | –0.41 |
| *Lyonia mariana* | Native | 68 | 33 | –0.75 |
| *Magnolia virginiana* | Native | 18 | 16 | –0.42 |
| *Malus coronaria* | Native | 8 | 8 | –0.68 |
| *Malus pumila* | Introduced | 13 | 22 | 0.49 |
| *Menispermum canadense* | Native | 48 | 42 | 0.12 |
| *Morus alba* | Introduced | 20 | 81 | 2.41 |
| *Morus rubra* | Native | 20 | 8 | –1.71 |
| *Myrica gale* | Native | 34 | 13 | –1.52 |
| *Myrica pensylvanica* | Native | 112 | 63 | –0.22 |
| *Nemopanthus mucronatus* | Native | 25 | 10 | –1.60 |
| *Nyssa sylvatica* | Native | 62 | 75 | 0.88 |
| *Ostrya virginiana* | Native | 46 | 39 | 0.03 |
| *Parthenocissus quinquefolia* | Native | 58 | 84 | 1.19 |
| *Parthenocissus vitacea* | Native | 7 | 7 | –0.75 |
| *Paulownia tomentosa* | Introduced | 8 | 18 | 0.69 |
| *Philadelphus coronarius* | Introduced | 10 | 16 | 0.24 |
| *Physocarpus opulifolius* | Native | 25 | 17 | –0.70 |
| *Picea rubens* | Native | 10 | 8 | –0.92 |
| *Pinus echinata* | Native | 8 | 6 | –1.15 |
| *Pinus rigida* | Native | 44 | 34 | –0.16 |
| *Pinus strobus* | Native | 33 | 33 | 0.13 |
| *Pinus virginiana* | Native | 20 | 7 | –1.93 |
| *Platanus occidentalis* | Native | 13 | 32 | 1.16 |
| *Populus alba* | Introduced | 11 | 13 | –0.22 |
| *Populus deltoides* | Native | 16 | 42 | 1.41 |
| *Populus grandidentata* | Native | 73 | 54 | 0.05 |
| *Populus tremuloides* | Native | 56 | 43 | –0.03 |
| *Potentilla fruticosa* | Native | 28 | 12 | –1.42 |
| *Prunus avium* | Introduced | 23 | 42 | 0.99 |

| | | | | |
|---|---|---|---|---|
| *Prunus maritima* | Native | 46 | 32 | –0.32 |
| *Prunus pensylvanica* | Native | 14 | 12 | –0.62 |
| *Prunus pumila* | Native | 18 | 9 | –1.39 |
| *Prunus serotina* | Native | 74 | 101 | 1.25 |
| *Prunus virginiana* | Native | 39 | 31 | –0.18 |
| *Ptelea trifoliata* | Native | 8 | 10 | –0.31 |
| *Pyrus communis* | Introduced | 6 | 10 | 0.00 |
| *Quercus alba* | Native | 57 | 77 | 1.04 |
| *Quercus bicolor* | Native | 49 | 46 | 0.26 |
| *Quercus coccinea* | Native | 40 | 49 | 0.62 |
| *Quercus ilicifolia* | Native | 70 | 42 | –0.35 |
| *Quercus marilandica* | Native | 32 | 25 | –0.32 |
| *Quercus montana* | Native | 48 | 53 | 0.54 |
| *Quercus muhlenbergii* | Native | 6 | 9 | –0.17 |
| *Quercus palustris* | Native | 31 | 53 | 1.07 |
| *Quercus phellos* | Native | 11 | 15 | 0.02 |
| *Quercus prinoides* | Native | 51 | 26 | –0.81 |
| *Quercus rubra* | Native | 50 | 78 | 1.23 |
| *Quercus stellata* | Native | 36 | 26 | –0.39 |
| *Quercus velutina* | Native | 60 | 76 | 0.95 |
| *Rhamnus cathartica* | Introduced | 16 | 22 | 0.26 |
| *Rhamnus frangula* | Introduced | 10 | 32 | 1.45 |
| *Rhododendron maximum* | Native | 28 | 28 | 0.04 |
| *Rhododendron periclymenoides* | Native | 94 | 56 | –0.21 |
| *Rhododendron viscosum* | Native | 105 | 59 | –0.26 |
| *Rhus copallinum* | Native | 55 | 51 | 0.30 |
| *Rhus glabra* | Native | 72 | 65 | 0.42 |
| *Rhus hirta* | Native | 44 | 48 | 0.46 |
| *Ribes americanum* | Native | 23 | 19 | –0.41 |
| *Ribes hirtellum* | Native | 24 | 8 | –1.92 |
| *Ribes rotundifolium* | Native | 20 | 16 | –0.54 |
| *Ribes rubrum* | Introduced | 18 | 24 | 0.28 |
| *Robinia hispida* | Introduced | 10 | 12 | –0.25 |
| *Robinia pseudo-acacia* | Introduced | 21 | 60 | 1.76 |

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|---|---|---|---|---|
| *Robinia viscosa* | Introduced | 19 | 6 | –2.13 |
| *Rosa carolina* | Native | 105 | 58 | –0.29 |
| *Rosa eglanteria* | Introduced | 16 | 7 | –1.67 |
| *Rosa multiflora* | Introduced | 14 | 79 | 2.79 |
| *Rosa palustris* | Native | 50 | 45 | 0.19 |
| *Rosa rugosa* | Introduced | 11 | 14 | –0.09 |
| *Rosa virginiana* | Native | 38 | 16 | –1.30 |
| *Rubus allegheniensis* | Native | 67 | 43 | –0.25 |
| *Rubus flagellaris* | Native | 49 | 34 | –0.29 |
| *Rubus hispidus* | Native | 48 | 35 | –0.21 |
| *Rubus laciniatus* | Introduced | 13 | 15 | –0.16 |
| *Rubus occidentalis* | Native | 46 | 37 | –0.06 |
| *Rubus odoratus* | Native | 48 | 19 | –1.29 |
| *Rubus pensilvanicus* | Native | 34 | 29 | –0.13 |
| *Rubus phoenicolasius* | Introduced | 35 | 62 | 1.22 |
| *Salix alba* | Introduced | 16 | 13 | –0.64 |
| *Salix bebbiana* | Native | 31 | 12 | –1.54 |
| *Salix discolor* | Native | 74 | 83 | 0.86 |
| *Salix eriocephala* | Native | 43 | 41 | 0.21 |
| *Salix fragilis* | Introduced | 12 | 11 | –0.60 |
| *Salix humilis* | Native | 84 | 19 | –2.01 |
| *Salix nigra* | Native | 42 | 63 | 1.03 |
| *Salix purpurea* | Introduced | 15 | 10 | –1.01 |
| *Salix sericea* | Native | 54 | 24 | –1.02 |
| *Sambucus canadensis* | Native | 74 | 79 | 0.76 |
| *Sambucus racemosa* | Native | 29 | 15 | –1.08 |
| *Sassafras albidum* | Native | 66 | 97 | 1.31 |
| *Smilax glauca* | Native | 69 | 44 | –0.25 |
| *Smilax rotundifolia* | Native | 59 | 67 | 0.73 |
| *Solanum dulcamara* | Introduced | 67 | 78 | 0.86 |
| *Spiraea alba* | Native | 64 | 51 | 0.12 |
| *Spiraea tomentosa* | Native | 63 | 33 | –0.65 |
| *Staphylea trifolia* | Native | 48 | 50 | 0.43 |
| *Symphoricarpos orbiculatus* | Introduced | 9 | 9 | –0.61 |

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|---|---|---|---|---|
| *Tilia americana* | Native | 35 | 57 | 1.06 |
| *Toxicodendron radicans* | Native | 45 | 54 | 0.65 |
| *Toxicodendron vernix* | Native | 30 | 31 | 0.13 |
| *Tsuga canadensis* | Native | 34 | 40 | 0.44 |
| *Ulmus americana* | Native | 36 | 59 | 1.09 |
| *Ulmus rubra* | Native | 35 | 41 | 0.45 |
| *Vaccinium angustifolium* | Native | 72 | 29 | –1.05 |
| *Vaccinium corymbosum* | Native | 159 | 87 | –0.11 |
| *Vaccinium macrocarpon* | Native | 66 | 21 | –1.51 |
| *Vaccinium pallidum* | Native | 104 | 70 | 0.08 |
| *Vaccinium stamineum* | Native | 64 | 46 | –0.07 |
| *Viburnum acerifolium* | Native | 108 | 82 | 0.33 |
| *Viburnum dentatum* | Native | 101 | 92 | 0.65 |
| *Viburnum lentago* | Native | 38 | 39 | 0.26 |
| *Viburnum nudum* | Native | 57 | 30 | –0.70 |
| *Viburnum opulus* | Native | 16 | 22 | 0.26 |
| *Viburnum prunifolium* | Native | 74 | 85 | 0.90 |
| *Viburnum rafinesquianum* | Native | 19 | 14 | –0.71 |
| *Vitis aestivalis* | Native | 82 | 66 | 0.28 |
| *Vitis labrusca* | Native | 81 | 69 | 0.38 |
| *Vitis riparia* | Native | 25 | 31 | 0.35 |
| *Vitis vulpina* | Introduced | 19 | 17 | –0.38 |
| *Zanthoxylum americanum* | Native | 12 | 20 | 0.42 |