INVESTIGATING THE IMPACTS OF VEGETATION ON FOOD WEB COMPLEXITY

by

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ABSTRACT

In light of the increasing environmental pressures of habitat loss and expanding human development, these studies sought to explore how changes in vegetation impact biodiversity and ecosystem function. My first study investigated the effects of invasive plant species in unmanipulated hedgerows on the abundance, richness, and biomass of Lepidoptera larvae. Invaded transects produced significantly fewer individual caterpillars, less biomass, and fewer species than did transects not heavily invaded by evolutionarily novel plants. These reductions were also much greater than those observed in controlled settings testing similar effects, indicating that the effects of invasive plants are compounded in hedgerows where uncontrolled colonization occurs. The second study examined how land management and vegetation structure influenced local breeding bird populations. I conducted point-counts within mowed turf sites, meadow sites, and forest sites, and compared avian species richness, diversity, and evenness among each land use type. I found that forested sites had significantly greater diversity and richness than both meadow and turf sites, demonstrating the reduced ability of turf grass to support nesting and foraging needs of breeding birds. These studies may have beneficial implications in encouraging land management techniques that promote reducing area devoted to turf grass (lawns) and using native plantings to support greater biodiversity.

Chapter 1

IMPACT OF INVASIVE PLANTS ON LEPIDOPTERA POPULATIONS
Introduction

In recent decades both natural and managed landscapes throughout much of the U.S. have been altered by the introduction of non-native plants (Mckinney 2001; 2006; Standley 2003; De Candido et al. 2004). These species have been introduced either accidentally or deliberately for use as ornamental landscape plants in locations other than where they evolved. Many introduced plants invade native plant communities and create evolutionarily novel plant communities (Hobbs et al. 2009). Such invasions have been shown to influence the biotic and abiotic structure and function of the invaded community in several ways. However, one potential consequence of the wide-scale replacement of native plants with introduced plant species that has yet to receive sufficient experimental attention is how an altered first trophic level impacts insect herbivores (Tallamy 2004).

Ecological specialization is the rule rather than the exception (Forister et al. 2012), and this is particularly true of insect herbivores. An estimated 90% of herbivorous insect species are specialists, with their feeding and reproduction restricted to a few plant lineages at most (Ehrlich and Raven 1964, Bernays and Graham 1988, Novotny et al. 2006). Such specialization restricts feeding options,

rendering evolutionarily novel plants unavailable unless by relatedness or coincidence they also possess the defense to which the insect is adapted. This general pattern has been well-supported through the years (Farrell and Mitter 1990, Berenbaum 1990, Weiblen et al. 2006, Burghardt et al. 2010), but exceptions are surfacing that reflect how the plant defense is elicited, which, in turn, may reflect the feeding mechanism employed by the insect (Verhoeven et al. 2009).

Mismatches in toxin-based defensive interactions between plants and insect herbivores that are caused by the novel introduction of a plant often confer an advantage to the plant because local herbivores do not have the adaptations to detoxify novel defensive compounds (enemy release hypothesis; Keane and Crawley 2002). In contrast, mismatches in induced elicitor-receptor defense systems that occur when a novel member of a local plant lineage is introduced may provide an advantage to local specialist herbivores because they can feed on the novel plant undetected (biotic resistance hypothesis; Elton 1958, Verhoeven at al. 2009). Results of laboratory studies have supported both the enemy release hypothesis (Tallamy and Shropshire 2009; Tallamy et al. 2010, Burghardt et al. 2010) and the biotic resistance hypothesis, (Parker and Hay 2005) suggesting that how a particular community of insect herbivores responds to novel host choices may be nuanced by the plant and insect species within the community in question, and/or by the feeding mode of the insects.

Recently Burghardt et al. (2010) used a large replicated common garden experiment to show that, under controlled conditions, introduced plants that were congeners of local native plants reduced Lepidoptera species richness and abundance

on average 50%. Introduced plants that were unrelated to any local native plants reduced Lepidoptera richness and abundance by 75%. Here, we compare the Lepidoptera community in invaded and uninvaded hedgerows to examine how moths and butterflies respond to unmanipulated changes to local native plant communities. If the most pervasive plant invaders are particularly unpalatable to native insect herbivores, the impact on insect communities and the insectivores dependent on them may be greater than measured in controlled experiments. We chose Lepidoptera to facilitate comparisons between our study and that of Burghardt et al. (2010) and because lepidopteran larvae (caterpillars) are disproportionately valuable sources of food for many terrestrial birds, particularly warblers and neotropical migrants of conservation concern (Morse 1989, Dunn & Garrett 1997). Understanding their response to alterations in the first trophic level thus has important conservation implications.

Methodology

Study Sites and Survey Methods

Surveys were conducted in New Castle County DE hedgerows that were either heavily invaded by non-native plants or were largely native in plant species composition (hereafter invaded and uninvaded sites). Surveys were conducted twice at each site: once in June 2011 and once in July 2011, and only on days when foliage was not wet. At each site, we surveyed caterpillars along 100 meter transects that followed the hedgerow edge. We focused on edge habitat because of logistical ease and because leaves exposed to the sun support more Lepidoptera larvae than do leaves in shade (Levesque et al. 2002, Lincoln & Mooney 1984). Uninvaded transects, defined as habitats composed of ≥95% native plants by biomass, served as control habitats; invaded sites were defined as habitats composed of ≥75% non-native plant biomass. For each sampling period, eight unique sites were sampled: four uninvaded and four invaded. Sites were separated from each other by at least 100 meters but all sites occurred within 5 kilometers of each other, all were within the piedmont of mid-Atlantic deciduous forest, and thus all were exposed to similar populations of potential Lepidoptera colonizers.

For sampling we used the total search method (Wagner 2005) to methodically inspect leaves, twigs and branches of woody plants from the ground to 2 meters high,

every 2 meters along each 100 meter transect for 5 minutes. Thus, we searched each

treatment for a seasonal total of 2000 minutes. All larvae encountered were measured

in millimeters, their host plant was recorded, and they were identified to species or

morpho-species. Larvae that could not be identified in the field were kept to be

identified in the lab using the literature and the University of Delaware Insect

Reference collection. When needed and possible, larvae were reared to adulthood for

identification. We estimated plant biomass at each sampling stop by evaluating the

relative percentage of space occupied by each plant species within the $2m^3$ area being

searched. Larval biomass was estimated from measurements of larval length using the

formula derived by Rogers et al. (1976).

$$W= 0.0305 \ L^{2.62}$$
where W is weight in mg and L is length in mm.

Data Analysis

Mean larval abundance, species richness, and biomass were expressed per

units of plant biomass and calculated as a seasonal total for each treatment.

Treatments were compared by *t*-test. We calculated a Lepidoptera productivity index

for each hedgerow by multiplying the number of Lepidoptera species known in the

literature to utilize each plant species searched by the percentage of space each of

these plants species occupied within the $2m^3$ areas searched along each transect. This

index allowed us to quantify the potential productivity of each transect given the

species composition and density of plant biomass it contained. Lepidoptera host

records were obtained from Tallamy and Shropshire (2009).

Results

Plant Surveys

Although invaded hedgerows did not differ from uninvaded hedgerows at the site level in plant species richness per sampling (2.54 ± SE/ $2m^3$ uninvaded vs. 2.2± SE/$2m^3$ invaded; $t = 1.39$ invaded; df = 13; $P = 0.185$) nor in plant biomass (70 ± SE/$2m^3$ uninvaded vs 62 ± SE/$2m^3$ invaded; $t = 2.04$; df = 13; $P = 0.0613$), they were significantly less productive in terms of generating Lepidoptera larvae than native hedgerows ($t = 3.66$; df = 13; $P = 0.0029$; Fig. 1a). Furthermore, plant species richness per transect was significantly higher in uninvaded transects (25.5 ± SE/$2m^3$) when compared to invaded transects (16.25 ± SE/$2m^3$; $t = 2.87$; df = 13; $P = 0.013$). A total of 73 plant species were recorded in uninvaded transects compared to only 45 species in invaded transects (Figure 2).

(a) Plant productivity index

(b) Larvae species richness

(c) Larval biomass
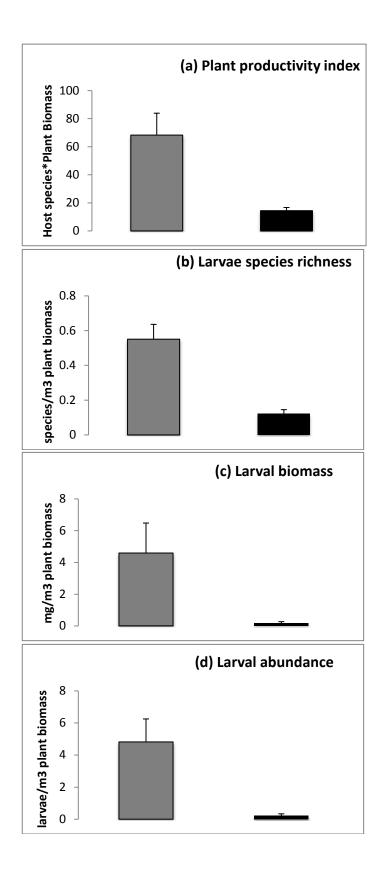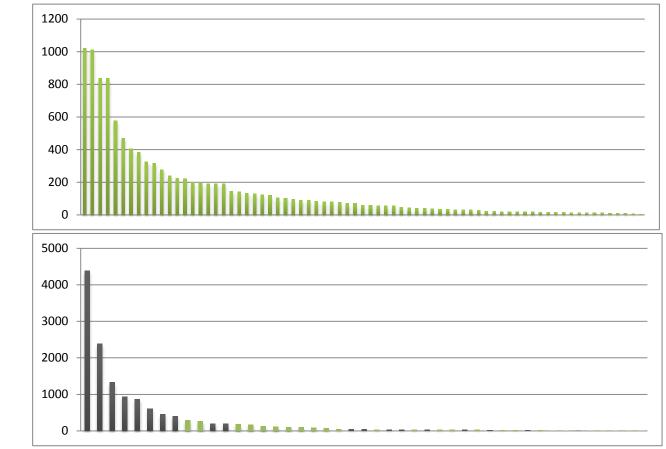
(d) Larval abundance

Figure 1) Uninvaded and invaded hedgerows compared in terms of Lepidoptera productivity index (a), larval species richness (b), abundance (c) and biomass (d). Statistical intervals = Standard Errors of the Mean.

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Figure 2) Cumulative plant compositions of Uninvaded (upper) and Invaded (lower) transects, with each bar representing a plant species. Native species are pictured in green, nonnative species are in black.

Larvae Surveys

Data collected from sampling Lepidoptera larvae were used to compare invaded and uninvaded transects in terms of three parameters: larval species richness, larval abundance, and larval biomass. Larval species richness was five times greater per unit plant biomass in uninvaded hedgerows than in invaded hedgerows (Fig. 1b). Unpaired two-sample t-test results showed this difference to be significant ($t = 5.15$; df $= 13$; $P = 0.0002$). Uninvaded hedgerows also contained significantly more Lepidoptera larvae per unit plant biomass than invaded transects ($t = 3.43$; df $= 13$; P

= 0.0045; Fig. 1d) as well as significantly more larval biomass ($t = 2.51$; df = 13; $P =$ 0.0259; Fig 1c).

Discussion

Our results show that under unmanipulated conditions, invasions of hedgerows by woody plants with no evolutionary history in the mid-Atlantic region of the U.S. result in a five-fold reduction in caterpillar species, a 22-fold reduction in caterpillar abundance, and a 23-fold reduction in caterpillar biomass compared to hedgerows dominated by indigenous woody plant species. These losses of caterpillar diversity and abundance are greater than those observed within the controlled common garden experiments of Burghardt et al. (2010), in which caterpillar species richness and abundance were reduced 75% on novel plants with no local native congeners. While both studies demonstrate a serious impact on caterpillar-based food webs when nonnative plants replace indigenous plant species, our comparison of invaded and uninvaded hedgerows reveal a greater loss of caterpillars due to the actual plant compositions of invaded hedgerows. Sampled invaded hedgerows were composed not of a random collection of non-native plants, as was the case in the common garden experiment, but rather a collection of aggressively invasive woody plant species. These species included autumn olive (*Eleagnus umbellata*), which dominated most transects, followed closely by multiflora rose (*Rosa multiflora*), bush honeysuckle (*Lonicera maackii*), Oriental bittersweet (*Celastrus orbiculatus*), and porcelainberry (*Ampleposis brevipedunculata*) (Table 1). These species, as well as the less abundant

Japanese honeysuckle (*Lonicera japonica*), bamboo (*Pseudosasa japonica*), and Japanese knotweed (*Fallopia japonica*) are exceptionally poor host plants for larval Lepidoptera, supporting few to no species under natural invasion field conditions (Tallamy and Shropshire 2009).

Larval Lepidoptera are a primary food source for many insectivores, particularly terrestrial birds (Morse 1989, Dunn & Garrett 1997), and insect herbivores are known to have a critical role in transferring energy from plants to other trophic levels (Elton 1927). The drastic reduction in the availability of this prey item in invaded habitats indicates the reduced ability of these areas to support productive connections between species at numerous trophic levels.

These results are critical to further understanding the large-scale effects of nonnative plants not only in unmanipulated settings but also within managed urban and suburban areas. Landscaping practices have increasingly favored introduced ornamental plants, especially those with acknowledged abilities to repel herbivorous insect "pests" (Dirr 1998, Tallamy 2004). In the past century, managed landscapes have become dominated by evolutionarily novel plant species (Hobbs et al. 2009); our study suggests that these nonnative ornamentals may be unable to support stable, complex food webs. With biological diversity currently threatened by a variety of pressures, particularly habitat loss, it is important to address the effects of nonnative plant species in both managed and unmanaged landscapes in terms of their subsequent impacts on higher trophic levels. Loss of diversity and abundance at the second trophic level (herbivores) has been shown to destabilize higher trophic levels and limit

ecosystem function (Schmitz et al. 2000, Mulder et al. 1999). These advances in our understanding of how nonnative plants impact ecosystems may have important conservation implications but also may aid in shifting the cultural perception of plants as merely decorative objects.

Chapter 2

AVIAN DIVERSITY AMONG LAND USE TYPES

Introduction

Ecosystem services comprise a multitude of beneficial processes such as nutrient cycling, water purification, and pest management that are supplied by functioning biological communities (Tilman et al. 2006, Cardinale 2011). Complex food webs with an abundance of potential trophic connections have been shown to be more stable, higher functioning, and possess greater resistance to environmental disturbance and fragmentation (Montoya and Sole 2003). As part of a larger project exploring the relationships between landscape management, water quality, and ecosystem services, I investigated the ability of several landscape management approaches to support avian populations. Bird populations contribute to the maintenance of arthropod populations at appropriate levels and therefore reduce the need for pesticides as well as assist in preventing the spread of insect-borne disease. During the breeding season, birds rely heavily on Lepidoptera larvae to feed their young, and are consequently an indicator of the status of caterpillar abundance and diversity as well. Furthermore because birds are a taxon both popular with the general public, as well one with many species experiencing decline, they are a group of particular interest (North American Bird Conservation Initiative 2011).

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Due to the biological control services provided by birds, their populations are also beneficial in reducing the need for chemical pesticides and thus improving water quality. Water plays a critical role in sustaining organisms and therefore pollution of waterways has major impacts on ecosystem health, as well as human health (Bozoglu 2011). Within urbanized watersheds, the conversion of landscape to turf grass and the subsequent use of fertilizers and pesticides is a common source of pollution (Schueler 1994). Previous research has shed light on the effects on water quality when the vegetation abundance and diversity of a watershed is enhanced (Omernik 1976; Smart et al. 1985; Pollock and Meyer 2001), however the effects of adding vegetation to watersheds for other ecosystem services have yet to be fully explored. Quantifying the ability of different landscape management practices to support breeding bird populations will offer further information on the effects of mowing and vegetation structure on the presence of biological pest control services and ecosystem stability.

Methodology

I conducted breeding bird surveys in three land management types to assess avian presence as an indicator of pest control as well as a measure of biodiversity and ecosystem health. I used 25-meter, fixed-radius point-counts (Donnelly and Marzluff 2004) to estimate breeding bird presence on the grounds of Winterthur Museum and Gardens in Wilmington DE, where I selected two sites each of three land use types (forest, meadow, and mowed turf) for a total of six survey locations. All surveys occurred from June 11 to July 24 in 2012 between 6 and 11 am, at air temperatures below 95° F (35° C), and during periods of no precipitation. Ten point-counts were conducted at each site and a minimum of three days elapsed between each survey. I recorded all native birds seen or heard within a 25-meter radius of each point within a ten-minute period. I estimated avian abundance at each site by summing the maximum number of individuals detected across the point counts for each species and dividing it by the number of points sampled on a property.

The turf plots were composed of mostly tall fescue (*Festuca arundinacea*) and were mowed once weekly throughout the growing season. Meadow plots were also predominantly tall fescue but only mowed twice per season. Forest plots were mature hardwoods consisting of white oak (*Quercus alba*), red oak (*Q. rubra*), hickory (*Carya spp.*), beech (*Fagus grandifolia*), and several species of woody understory trees and

shrubs. Comparisons of avian species richness were made with the land use type as the

sampling unit ($n = 6$) and compared by ANOVA. Mean avian species evenness and

diversity levels for three habitat types were determined using Shannon's diversity

index

Results

Avian species richness, evenness, and diversity compared between mowed turf, meadow, and forest are shown in Figure 3. There was a significant difference in species richness among all land uses (one-way ANOVA $F_s= 26.292$, p= 8.12×10^{-9}), with forested sites possessing the greatest average number of species at 7.8, followed by meadow areas (5.4) and turf (3.9). Species evenness was significantly lower in forest sites than in turf sites ($F_s= 8.836$, p= 4.54×10^{-4}). Differences in species diversity were significant among all land use types ($F_s= 40.124$, p= 1.33×10^{-11}), with forested sites having greater diversity on average than meadow and turf sites.
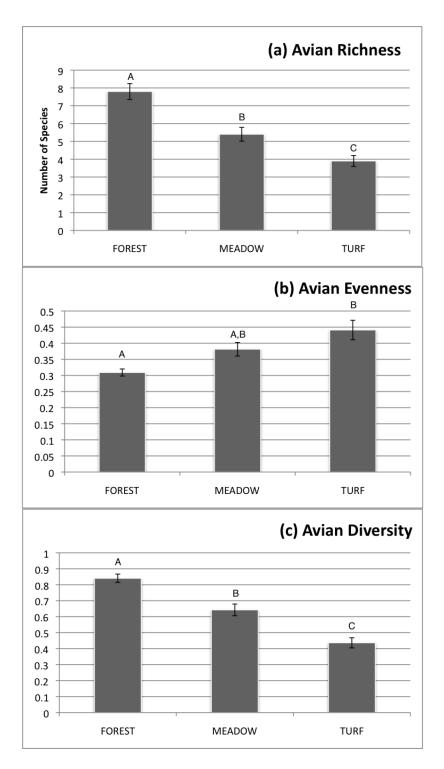
.

Figure 3) Mean estimates of Avian Species Richness, Evenness, and Diversity. Error bars indicate +/- standard error. Means with the same letter are not significantly different from each other (Tukey–Kramer test, $P<0.05$).

Discussion

The results of these surveys illustrate the effects vegetative structure can have on breeding bird populations. If watershed management uses plants to reduce water pollution by reducing areas of turf grass, valuable breeding bird habitat can also be created. While the forested sites showed both the greatest species richness and diversity, meadows also had more richness and diversity than regularly mowed turf grass. Evenness showed the opposite trend, with turf displaying the greatest overall evenness followed by meadow and forest sites. This was likely due to the more homogeneous conditions in turf sites, where species richness as well as individual abundance were extremely consistent across samplings. Bird species observed in mowed turf plots typically were tree swallows (*Tachycineta bicolor*), eastern bluebirds (*Sialia sialis*), and red-winged blackbirds (*Agelaius phoeniceus*) (Table 2). Swallows and bluebirds were present due to the availability of numerous nest boxes stationed throughout turf sites, and red-winged blackbirds were common near the several creeks running through the expansive turf areas, however no birds actually nested in the turf.

The absence of many birds in areas of reduced vegetative complexity reflects not only a lack of suitable nesting habitat but also, a lack of support for the insect food sources, particularly caterpillars, necessary to support breeding populations. Most host plants for Lepidoptera are conspicuously absent in turf areas, and in low abundance in meadows. The loss of birds in mowed areas displays the ecological paucity resulting

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from management of landscapes that reduces and simplifies vegetative structure. In light increasing habitat loss and expanding human development, the need to reexamine how human-dominated landscapes are constructed and managed is critical.

Our results may under-represent the conditions for birds that exist in many suburban areas due to the proximity of the Winterthur turf sites included in this study to meadows and forests, as well as a small stream. Truly isolated plots of turf grass, as they exist in many urban, residential, and industrial areas, typically do not have biologically rich habitats nearby that can help support species' foraging within mowed areas. In reality, the ability of much of the turf grass in the U.S. to support biodiversity is likely to be even more reduced than measured here. If simplified landscapes were uncommon, these impacts could be nominal; however, manicured lawns have become the norm and this type of landscaping covers an estimated 45.6 million acres, mostly concentrated in the eastern U.S. (Robbins and Birkenholtz 2003).

CONCLUSIONS

In each of these projects, I sought to investigate how a habitat's vegetation affected its ability to support biodiversity and, consequently, stable and functioning food webs. My results from Lepidoptera surveys illustrated the low level of productivity of many nonnative plants, and my avian point-counts showed how vegetative structure and land management practices impact breeding bird populations. These studies were not conducted in residential landscapes, but the results have valuable implications for landscape management in urban and suburban environments. Both studies illustrate the importance of diverse, native plant communities in supporting biodiversity and thus ecosystem function.

Human use and perception of plants remains disconnected from the many critical ecological functions of plants. Monocultures of lawn and a propensity for exotic ornamentals in landscaping are still commonplace practices that result in ecologically depauperate habitats, void of many of the services that a functioning landscape could provide. In these unsustainably managed areas, biodiversity, pest control, pollination, carbon sequestration, water filtration, and other ecological processes are compromised.

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APPENDIX

Table 1: Plant Compositions of Invaded and Uninvaded Transects

Invaded Transects			% of Total		Native Transects
Species	Biomass/m^3			Biomass/m^3	Species
Autumn Olive	4387	31.599	9.416	1021.5	(*Vibernum dentatum*)
Multiflora Rose	2390	17.215	9.310	1010	(*P. serotina*)
Bush Honeysuckle	1327.5	9.562	7.715	837	American Beech
Celastrus	930	6.699	7.701	835.5	Red Maple (*Acer rubrum*)
Porcelain Berry	863.5	6.220	5.309	576	Sweetgum
Bamboo	607.5	4.376	4.332	470	White Oak
Japanese Honeysuckle	455	3.277	3.733	405	(*Nyssa sylvatica*)
Japanese Knotweed	402.5	2.899	3.526	382.5	Tuliptree
Spicebush	290	2.089	3.014	327	Black Oak
Sassafras	270	1.945	2.913	316	Hickory
White mulberry	195	1.405	2.558	277.5	Carpinus
Paulownia	192.5	1.387	2.212	240	Slippery elm
Black cherry (*Prunus serotina*)	182.5	1.315	2.069	224.5	Red oak
Boxelder	177.5	1.278	2.028	220	Pin oak
Winged sumac	132.5	0.954	1.843	200	(*Cornus florida*)
Tuliptree	110	0.792	1.797	195	White Ash
Black Walnut	107.5	0.774	1.751	190	Black Walnut
Red Maple (*Acer rubrum*)	97.5	0.702	1.751	190	Persimmon
(*Cornus florida*)	82.5	0.594	1.751	190	Scarlet Oak
White Ash	72.5	0.522	1.337	145	Sycamore
Hickory	50	0.360	1.313	142.5	Spicebush
Mimosa	46.5	0.335	1.221	132.5	(*V. prunifolium*)
Privet	42.5	0.306	1.198	130	(*C. alternifolia*)
Sweetgum	40	0.288	1.129	122.5	American Plum
American Holly	37.5	0.270	1.106	120	American Chestnut

Name					Name
Amur Honeysuckle	37.5	0.270	0.968	105	Blueberry
Smilax	36.5	0.263	0.945	102.5	Chestnut Oak
Norway Maple	35	0.252	0.876	95	Shingle Oak
American Beech	32.5	0.234	0.830	90	(*Ilex verticillata*)
Silver Maple	32.5	0.234	0.830	90	Winged Sumac
(*Malus spp.)* (nonnative)	27.5	0.198	0.760	82.5	Black Willow
Red oak	27.5	0.198	0.737	80	Buttonbush
Redosier dogwood	25	0.180	0.737	80	Pawpaw
(*Malus spp.)* (native)	20	0.144	0.714	77.5	Sassafras
Native grape	20	0.144	0.659	71.5	(*V. acerifolium*)
Asian pear	17.5	0.126	0.645	70	Dwarf Chestnut
(*Vibernum prunifolium*)	17.5	0.126	0.553	60	White pine
Persimmon	12.5	0.090	0.553	60	American Elm
Red Cedar	12.5	0.090	0.530	57.5	Black Birch
Buckthorn	10	0.072	0.507	55	River Birch
Filbert	10	0.072	0.507	55	(*Pinus virginiana*)
Smooth Sumac	10	0.072	0.438	47.5	Washington Hawthorne
Black Oak	5	0.036	0.415	45	Red Cedar
Sycamore	2.5	0.018	0.369	40	Gray Dogwood
Witchhazel	2.5	0.018	0.369	40	Shadbush
			0.346	37.5	Fringe tree
			0.323	35	Crabapple
			0.323	35	Sugar Maple
			0.300	32.5	(*Celtis occidentalis*)
			0.300	32.5	Silverbell
			0.300	32.5	Witchhazel
			0.253	27.5	Poplar
			0.207	22.5	Smilax
			0.207	22.5	Turkey Oak
			0.184	22.5	(*Clethra alnifolia*)
			0.184	20	Filbert
			0.184	20	Loblolly Pine
			0.184	20	(*Ostrya spp.*)
			0.184	20	American Willow (*Salix discolor*)
			0.161	17.5	(*Aronia spp.)*
			0.138	15	Alder
			0.138	15	Blackjack oak

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			0.138	15	Gray Birch
			0.115	12.5	Boxelder
			0.115	12.5	Hawthorne
			0.115	12.5	Redbud
			0.115	12.5	Smooth Sumac
			0.115	12.5	Willow Oak
			0.092	10	Black Chokecherry
			0.092	10	Hearts a burstin (*Euonymus americanus*)
			0.092	10	Virginia creeper
			0.069	7.5	Green Ash
			0.046	5	Fox Grape (V*itis labrusca*)
Total	13883.5	100	100	10849	**Total**

Table 2:   Bird Species Observed Within Turf, Meadow, and Forest Sites

Forest	Meadow	Turf
American Robin	American Robin	American Robin
American Crow	American Crow	American Crow
Blue Jay	American Goldfinch	Barn swallow
Carolina Chickadee	Baltimore Oriole	Chipping Sparrow
Carolina Wren	Barn Swallow	Eastern Bluebird
Common Grackle	Brown-headed Cowbird	Red-tailed Hawk
Downy Woodpecker	Carolina Chickadee	Red-winged Blackbird
Eastern Phoebe	Chipping Sparrow	Song Sparrow
Eastern Towhee	Eastern Bluebird	Tree Swallow
Eastern Wood Pewee	Eastern Towhee	Turkey Vulture
Gray Catbird	European Starling	Yellow-shafted Flicker
Morning Dove	Field Sparrow	
Northern Cardinal	Gray Catbird	
Northern Mocking bird	Great Blue Heron	
Northern Parula	House Finch	
Ovenbird	Morning Dove	
Pileated Woodpecker	Northern Cardinal	
Red-bellied Woodpecker	Northern Mockingbird	
Red-eyed Vireo	Red-tailed Hawk	
Red-tailed Hawk	Red-winged Blackbird	
Scarlet Tanager	Song Sparrow	
Song Sparrow	Tree swallow	
Tufted Titmouse	Turkey Vulture	
White-breasted Nuthatch	Yellow-shafted Flicker	

Wood Thrush

Yellow-shafted Flicker