THE PHYTOGEOGRAPHY OF VACANT LOTS IN BALTIMORE, MD, USA: A TEST OF ISLAND BIOGEOGRAPHY

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Abstract. Analyses of urban ecosystems are steadily providing novel perspectives to the environment within and around urban areas. Prior analysis of plant species richness in urban vacant lots has suggested that they act as "islands in space and time amidst a sea of impervious surfaces" (Crowe, 1979, p. 1). Considering vacant lots to be island analogs places the vegetative species diversity of vacant lots within the framework of the equilibrium theory of island biogeography. The abundance of vascular plants within thirty-one vacant lots was sampled in a nine-block-square area in the neighborhood of Harlem Park in west Baltimore, MD, USA. Regression analysis indicated no significant relationship between species richness or species diversity with isolation. There was a significant negative relationship of species richness with area ($R^2 = .2$, $p=.02$) suggesting that the theory of island biogeography does not adequately explain the species variation within this area of vacant lots. Canonical correspondence analysis showed that clustering of species within the lots was partially driven by the variables of area, isolation, lot shape, species diversity and species richness. Within individual lots, the vegetational composition differed between the footprint of the demolished building and the relic backyard. Overall species richness and diversity may depend on a variety of factors that include but are not limited to the variables measured in this study. Other likely important factors not measured in this study include human management, seed dispersal rates and species imported for lot recovery.

INTRODUCTION

The diverse spatial mosaics of metropolitan areas provide an array of ecological situations in which to examine plant community structure in an environmental patch mosaic (Pickett et al., 2001). Researchers measuring this spatial heterogeneity have developed several classifications for urban vegetation. As one of the first American ecologists to conduct research in urban landscapes, Forest Sterns (Sterns 1971) classified spatial heterogeneity by looking at three vegetation types- ruderal, managed and residual. Other classification systems attempt to characterize natural habitats within urban systems by categorizing based on life forms (Rogers and Rowntree, 1988). Plant community studies like these, which have been largely descriptive in nature, present spatial heterogeneity as a factor influencing species richness (Pickett and others, 2001).

A framework which has been used to study spatial heterogeneity of vacant lots is the equilibrium theory of island biogeography (MacArthur & Wilson, 1967). Crowe (1979) did an analysis of urban vacant lots in Chicago that tested three predictions of island biogeography: (1) Rate of colonization will level off over time, (2) There is a positive relationship between species richness and island area, (3) There is an inverse relationship between species richness and distance from the source of propagules. Crowe determined that age was correlated with species richness up to an age of 40 months. Beyond that age, species richness did not increase, but was positively related to lot area and negatively related to lot isolation. This suggested that vacant lots have immigration-extinction rates similar to those of oceanic islands with other lots being significant sources of colonists.

Within a vacant lot, habitat differences may affect immigration and extinction rates in ways that extend beyond the effects of area and isolation. Urban backyards have been shown to provide habitat for the recruitment and establishment of various species of above-ground vegetation and insects. It has been suggested that augmentation of backyards can enhance the capacity of urban systems to provide appropriate habitat for organisms within a difficult matrix for many species (Sperling and Lortie, 2009; Rudd and others, 2002). Backyards were found to have structural diversity (i.e. trees and shrubs mixed with open grassy areas) that increased species aggregation in a confined area (Rudd and others 2002; Jules and Shahani 2003; Hodgkison and others, 2007). These diverse and unique habitats may differ between habitats that are managed on a city-wide scale, which can possess certain internal homogeneity with in the management regime (Haigh 1980).

In this study, I tested whether the theory of island biogeography applied to species richness in a nine block area of vacant lots in the west Baltimore, which had a collective area of approximately 9 km². The urban vacant lots in this study area are defined as a system of spatially discrete units that had vegetation height above 6 inches (15.24 cm) and was unoccupied by built structures. The 6 inch vegetation limit filtered out lots whose vegetation had had no time for recovery after mowing. The lots were linked by City-owned parks located in the center of every block (Fig. 1). The parks were mowed more frequently than the vacant lots, according to informal conversations with neighborhood residents and city officials in July, 2010. Since the lots are older than 40 months (City of Baltimore 2004), I worked under the assumptions that the lots had reached immigration-extinction equilibrium, area and isolation would have significant impact in determining this equilibrium and age would not be a confounding variable. Under these assumptions, I made the following predictions:

1. Species richness would be positively correlated with lot area.
2. Species richness would be inversely correlated with isolation.

I also tested the hypothesis that there would be distinct clusters of species within individual lots as well as within the system of lots themselves, due to compositional patterns not encompassed by the theory of island biogeography theory. I predicted that these clusters of species could provide insight to environmental factors that drive species distributions in vacant lots and would vary by species. This test was done to understand data not reflected by the island biogeography theory test, in addition to understanding the differences of habitat preference between species. Environmental factors of interest would include the effects of isolation and area on individual species in relation to other species in addition to the overall shape, species richness and diversity of the lots in relation to individual species. The compositional differences between the vegetation on the former building footprint and the relic backyards were also analyzed. I predicted that the two areas would have a different plant community due to habitat and history differences.

METHODS

Site Description and History

The study site for this research was the neighborhood of Harlem Park in the City of Baltimore, Maryland. The City of Baltimore is one of the oldest industrial cities in the United States and has been characterized as a "city of neighborhoods" (DiChiro, 2002). It displays pre-World War II urban spatial configuration with a monocentric form and high-density, affluent neighborhoods at the center, surrounded by high-density, low-income neighborhoods, and then mid-income less dense neighborhoods (Troy and others, 2007) although pockets of wealthy neighborhoods have persisted and emerged over the years. Over the last 50 years, the city has undergone extensive demographic and economic changes with its population declining 33% from nearly 950,000 in 1950 to its current population of 637,418 (Burch and Grove, 1993; US Census Bureau 2010). Widespread abandonment and disinvestment have occurred in several neighborhoods, particularly high-density, low-income neighborhoods (City of Baltimore, 2006). Currently, there are nearly 30,000 abandoned properties (structures and lots) in the city, which is approximately 13% of the total property parcels in the city (Graziano and others, 2007). Of these abandoned properties, 11,198 are vacant lots without built structure, representing 5% of the total property parcels in the City.

Harlem Park neighborhood is a high-density, low-income neighborhood located in west Baltimore with a density 32.9% higher than the city's average and an average income 42.1% lower than the City's average (City-Data.com, 2008). For the purpose of this study, a nine-block square area was chosen between Lanvale Street and Franklin Avenue to the north and south, and Calhoun and Arlington Streets to the west and east (Figure 1). This area represented the typical neighborhood structure of occupied and unoccupied row houses, with sporadically located vacant lots composed of one to five property parcels and making up circa 7 % of the study area. The interiors of each block were “Inner Block Parks” (Harlem Park Project II, 1960), which were designed in an urban renewal project from the late 1950s and are currently minimally maintained and have deteriorated from their initial state. A catalogue of species from these parks has not been developed. However from personal observations of the study area, the Inner Block Parks were characterized by a relatively homogeneous composition of mown grass, trees such as white mulberry (*Morus alba* L.), princess tree (*Paulownia tomentosa* (Thunb.) Siebold & Zucc. ex Steud.), honey locust (*Gleditsia triacanthos* L.) and elm (*Ulmus parcifolia* Jacq., *Ulmus* sp. and *Zelkova serrata* (Thunb.) Makino.), some asphalt pavement and in a few cases, cement playground equipment (Appendix 1-1).

Sampling methods

The flora of 31 sites, representing all vacant lots in the study area as defined by the exclusion rule, was surveyed in July, 2009 (Figure 1). According to local residents, lots were mown occasionally by the city or by neighboring residents but not on a regular basis. Evidence of disturbance included footpaths, refuse dumps, parked cars, and dumped sand (Appendix 1-2). Four lots were removed because recent mowing (shorter than 6 inches) and three lots were fenced gardens. The lots with sand and gravel were not included in the analysis (*n* =2).

Vacant lots were composed of single or multiple vacant parcels. For each lot, a single parcel was chosen to sample based on a city map of parcel boundaries (Figure 2 and Appendix 1-3). Parcels were surveyed using a stratified random sampling technique. One m² quadrat frames were randomly placed within five meter strata along two transects extending the length of the parcel. Parcels were standardized by the measurement of 13’ by 72’, which were the typical dimensions of property parcels in the Harlem Park neighborhood. The building footprints were the areas of the lots which were once occupied by built structure and replaced by fill. They were represented by the quadrats of the first three strata. Relic backyards were assumed to have remained relatively intact throughout the demolition process and were designated by the backmost two quadrats.

Plants were identified to species, and prevalence was measured by percent coverage within the quadrat. Basal area of all trees over 5 cm in circumference within the lot was measured. Any species that could not be readily identified during the field survey was collected for a more thorough inspection in the laboratory. References that proved useful for species richness identifications were Uva and others (1997), Gleason & Cronquist (1991) and Holmgren (1998).

Variables

Variables tallied from field collections included species richness and the Shannon Weiner species diversity index. In ArcGIS using ISTAR® aerial photos (City of Baltimore 2004) along with Baltimore City property parcel layers (City of Baltimore 2007) quantitative variables were measured for each lot, which included area of lot in square meters distance to the next nearest vacant lot, area of lots within a 0.5 block buffer proportional to the different sizes of each block, distance to the largest lot in the sample, and length-to-width ratio of each lots. The range of the half-block buffers was 73m – 85m. The variables were chosen due to the following assumptions (Crowe, 1979): (1) Area is a measurement of propagule target size in space and (2) Distance to the nearest lot, area of the lots within a 0.5 block buffer, and distance to the largest lot within the sample are measurements of isolation from potential sources of propagules. The length-to-width ratio was used as a measure for lot shape. Variables and their abbreviations are summarized (Table 1). Unfortunately, age could not be measured as a variable due to time limitations and decentralized data in city records. This may be a significant shortcoming, as previous studies showed that age can be used to measure propagule target size over time (Crowe, 1979).

Statistical methods

Simple regressions using JMP® 8.0 (SAS Institute Inc. 2009) were done to test the relationship between the dependent variables, species richness and Shannon Wiener Diversity Index, and the independent variables of area, distance to the nearest lot, area of lots within 0.5.-block buffer, distance to the largest lot and length-to-width ratio. Unpaired t-tests were conducted between species diversity indices within the building footprint (Quadrats 1-3 and 5-7) and intact backyard parcels (Quadrats 4 & 8) ANOVA was also done to measure the relationship between species richness and lot area class (0 m^2-200 m^2, 200 m^2- 460 m^2- and -460 m^2- 730 m^2).

To describe gradients of species distribution, canonical correspondence analysis (Ter Braak, 1986) was applied using XLSTAT® (Addinsoft 2010). A canonical correspondence analysis (CCA) relates community variation to environmental variation. Environmental data included the quantitative variables of Shannon Wiener diversity index for each lot, species richness for each lot, area, distance to the nearest lot, area of lots within 0.5.-block buffer, distance to the largest lot and length-to-width ratio. These environmental variables distinguished the sampling success in measuring drivers of species variation by calculating constraint and unconstraint of the data that could be explained by the variables. A CCA diagram shows the "centers" of the species distributions along each of the environmental variables by accounting for the majority of variation within the top two factors (represented by the axes). The species distributional variance along environmental variables is represented by the length of the arrows.

RESULTS

The 117 flowering plant species identified are listed in the Appendix 2-1 (Table 2). Twenty species taxa could not be identified species. These twenty species were uncommon, being found on average 1.5 times in each lot and having an average coverage of 1.4% within the lots.

Within the lots, differences between the Shannon Wiener diversity index of the relic backyards and the building footprint were not significant (p=0.41). Of the 117 species present in the lots, 31 species were found only in the building footprint and 15 were found only in the relic backyard area (Table 3). This leaves the remaining 71 species having overlapping ranges in both habitat types. The top five species of the backyard section were *Hedera helix* L., *Schedonorus arundinaceus* (Schreb.) Dumort. Holub, *Dactylis glomerata* L., *Trifolium repens* L., and *Arctium minus* Bernh. The top five species in the building footprint were *Cynodon* dactylon (L.) Pers., *Schedonorus arundinaceus* (Schreb.) Dumort. Holub , *Dactylis glomerata* L., *Cichorium intybus* L., *Arctium minus* Bernh.

Of the 117 identified species, 21% were identified as native, while 47% were identified as invasive. The native range of the other species was unable to be identified. Exclusivity was calculated by a ratio of front abundance over back abundance. Ratios with zero as a numerator signified habitat exclusively in the front (Q1-3 and Q5-7) and ratios with zero in the denominator signified habitat exclusively in the back (Q4 and Q8). Of the known invasive species, 76% were located either primarily or exclusively in the building footprint, while 24% were located either primarily or mostly in the relic backyards. Of the known native species, this trend was reversed with only 46% occurring primarily or exclusively in the building footprint and 55% occurring in the relic backyard.

Simple regression analyses testing the relevance of island biogeography theory to the vacant lots showed insignificant or counterintuitive results (Table 4). Evidence of disturbances in the form of foot paths, refuse dumps, car parking and sand dumping, was factored out of regression analysis. Regression showed species richness to be significantly and negatively related with lot area (p=0.019) and characterized by the equation SPECIES RICHNESS = 23.688237 - 0.0129782AREA (Figure 3). Neither species richness nor Shannon-Wiener Species diversity indices were significantly correlated with distance to nearest lot, distance to the largest lot, length/width ratio or area of lots within the buffer.

Further analysis of the factors driving individual species variation within the lots was done using canonical correspondence analysis (CCA). In order to pick up the highest amount of variation, the analysis was limited to the top nine of the most abundant species throughout the sampled vacant lots, representing 58.8% of the total measured coverage in the sampled quadrats (Figure 4). The extent of individual species variation in relation to variation of measured variables was determined. These variables included lot area, all measurements of lot isolation (DISN, DISL and AW), species diversity of their respective lots (SWDI) and species richness of their respective lots (SPEC). Simulation with 1000 permutations indicated that the sites/objects data are not linearly related to the sites/variables data (*p*= 0.377). Constrained CCA accounted for 25.7% of the inertia, while results of the unconstrained CCA accounted for 74.3%. Eigenvalues gave a total of seven factors carrying the inertia (Figure 4). Sixty-six percent of the inertia is carried by the F1 and F2 axes. This suggests that while two dimensional CCA can be used to explain some of the variation between the top nine species, it should be recognized that other unmeasured variables also contribute to the variation.

The first and second axes in the CCA ordination (Figure 5) combine to account for 66% of the variation in species composition (Table 5). The regression coefficients can be used to measure the extent to which the variable contributes to the respective axis. A positive value signifies a positive influence and a negative represents a negative influence. For example, on the F2 axis species are primarily separated positively and negatively by the SWDI and the DISL. *Hedera helix* L. has a positive association with lots of high area and a negative association with lots with high species richness.

The first axis, which explains 33.5% of the plant species variance, contrasts species richness with area of lot. Specifically, *Dactylis glomerata* and *Arctium minus* are found in areas with relatively high species richness but relatively small lot sizes. Conversely, *Hedera helix* has relatively low species richness but relatively high lot area. The second gradient, showing 32.5% of the variance, contrasts the species diversity index with distance to the largest lot. Specifically, *Hedera helix* is found in areas with high isolation, as measured by DISL, and with low species diversity (SWDI), whereas *Plantago lanceolata* is low on the DISL scale but high on the SWDI. Of the total inertia, approximately 74% of the data was unconstrained. This means that of the 66% variation accounted for by the first two factors, only 26% of the data was constrained into the plot shown in Figure 5. However, this data still can be used to determine relationships of environmental variables.

Nineteen species of tree were encountered in the lots sampled (Table 6). Trees within the sample site were not abundant enough to determine spatial clustering and other forms of a statistical analysis. Ten of the 31 lots had trees greater than 5 cm in circumference. Half of these lots only had one species of tree.

DISCUSSION

Although there have been few studies analyzing the vegetative architecture of urban vacant lots, there have been studies that contextualize Harlem Park, Baltimore. The total species richness of the lot area (119 species or 95 species/km^2) was comparable to other urban land uses in previous studies, although exact relation is difficult to measure due to the fact that different studies employ different sample sizes. Crowe (1979) recorded a total of 128 plant species in the vacant lots of Chicago although he worked in a larger area with fewer and more isolated lots than the study area in Baltimore. In Plymouth, Devon, UK, analysis between macro-habitat types reported city parkland to have the highest vegetative species richness per area, with an average of 445 species/km^2, and recent housing development areas to have the lowest, with 15 species/km^2 (Kent and others, 1999). In areas surrounding the city of Paris, France, semi- and intermediate natural areas had the highest species richness while anthropogenic areas were usually species poor (Audrey and others, 2008). Wastelands had an average of 17 species/ha.

Compositional generalizations have often classified urban areas as having high abundances of common, ruderal and non-indigenous species that have been introduced intentionally and unintentionally by land managers, horticulturalists, gardeners and foresters over the historical period (Audrey and others, 2008; Kent and others 1999). This has created diverse urban insular environments that are relatively homogeneous at a global scale (McKinney 2005). Analysis of Baltimore lots support this generalization with a large proportion of exotic species classified as agricultural nuisances and garden escapes. There were some indigenous anomalies such as *Parthenocissus quinquefolia* (L.) Planch. and *Ageratina altissima* (L.) King & H. Rob. However, nine species with the highest coverage were all non-indigenous species originating in Europe, Asia or Africa.

Predictions of the equilibrium theory

Prediction 1: Species richness was inversely correlated with area. This contradicts the prediction based on the species area-relationship that is characteristic of the discrete habitats of the equilibrium theory of island biogeography. Although this prediction is not unique to the equilibrium theory (Wilson and Simberloff, 1969), positive correlation between area and species richness is characteristic of islands or island–like habitats (MacArthur and Wilson, 1967). A possible explanation for this occurrence however, could be explained by the management of the lots. In larger lots, the occurrence of mowing may play a role in determining homogeneity and conditions within the lot more so than in smaller, less visible lots. This contradiction may also be due to the possibility of multiple and unevenly distributed seed sources within and surrounding the sample area (Gilbert 1980), such as the case in the Baltimore lots. A third potential factor affecting the area-species richness relationship includes the introduction of species to lots that were seeded upon demolition of the building. Lots may have a different process of colonization after the demolition of the building depending on differing processes between demolition companies. Variance among lots of pioneer species may affect the succession process and the eventual equilibrium state of species richness.

Prediction 2. There was no statistically significant relationship between isolation and species richness. Theoretically the species richness should decrease with increasing isolation. However, this prediction fails when there is a possibility of more than one seed source. To overcome this, three tests of isolation (distance to the largest lot, distance to the nearest lot and area of other vacant lots within the buffer) were performed to account for possibilities of seeds traveling from one large single source or from surrounding areas. However, it is possible that seeds could be traveling from sources outside the measured landscape, such as from a large source outside of the study area, or from the interior parks found in the center of every block, or even from imported seed from mix or garden escapes. None of these variables were measured in the study and further investigation should be done to evaluate the impacts that these factors may have upon species distribution.

Species Clustering Hypothesis: Distinct clusters of species both within the individual lots and the system of lots as a whole were not significant. However, the tests may be seen as exploratory data for potential environmental factors that determine distribution within vacant lots of urban ecosystems.

The first part of this analysis was done by analyzing differences between the back and front parts of the lot. Although the difference between these two habitats was not significant, the test should be repeated using species richness instead of species diversity. Additionally, this variation within lots should be tested in CCA, by using front and back areas as dummy environmental variables.

The relic backyards and the building footprints showed distinctive species composition and origins of plants, but not collective species diversity. This may be due to the environmental and managerial differences between the two- areas from historic land use. Examples of differences could include gardens in backyards, shade differences between the two areas, or different substrate types due to fill that was recycled and/or imported during the demolition process. Finally, there is the possibility that soil used to cover the demolished soil included seed mix used to improve the soil by incorporating nitrogen fixers in the community.

Using CCA, relative differences of species distribution between the environmental variables within the lots in Harlem Park were found. While it should not be used to predict specific locations of individual species or environmental variables, the CCA could be important in recognizing the extent to which the environmental variables come into play.

Lots showed a high abundance and diversity of the Poaceae (11 species), Asteraceae (18 species) and Fabaceae (8 species) families. The top three species are perennial grasses, *Cynodon dactylon* (L.) Pers., *Dactylis glomerata* L., and *Schedonorus arundinaceus* (Schreb.) Dumort. Holuba, are introduced to the United States-. These species are considered invasive in many areas of the US. *Cynodon dactylon* (L.) Pers. (Bermudagrass) is a noxious weed in many areas due to its potential toxicity to livestock under certain environmental conditions. It also has a high production of pollen and can be a major cause of hay fever (USDA, NRCS 2010). The next three most abundant species were the forbs *Trifolium repens* L., *Arctium minus* Bernh. and *Cichorium intybus* L. Like the grasses, these species also are introduced to the US and are either perennial or biennial (USDA, NRCS 2010). All of the dominant species in these lots are often considered as weeds.

Many of these "weeds" however, also have many interesting characteristics. *Trifolium repens* L., for example, is a nitrogen fixer that assists the growth of grass in nutrient-poor soils. Historically, it has been planted with crop grasses for the production of hay. It can assist in erosion control. It is also highly palatable and provides nutritious forage for livestock (Gibbons 1971). *Cichorium intybus* L. has conspicuous blue flowers, is bee-pollinated and has a root that has been used as a non-caffeinated substitute for coffee. Even the most common species, *Cynodon dactylon* (L.) Pers., despite its noxious weed status in many areas, has had many historical medicinal uses (Gibbons 1971).

Assumption of age, management, and species equilibrium

In a previous study (Crowe, 1979), species richness and age based on demolition records, had a convex shape of species richness as a function of age. This represented an absence of statistical relation between species richness and age in lots older than 30 months (2.5 years). This indicates that older lots reached a point of species equilibrium, and thus, the 'snapshot' assumption can be made. This assumption presumes that the species richness of a young lot reflects the species richness of an older lot at an earlier stage in its colonization history. Crowe was able to justify this assumption with evidence showing that there was no strong effect of secondary succession (i.e. species that were found in younger lots were still found in older lots in addition to 41 new indicator species in the older lots. Younger lots only had 2 indicator species). Since all lots in Harlem Park were known to have been over 5 years of age (ISTAR 2004), according to Crowe's snapshot assumption, the age of the lots in this study should be irrelevant in determining the species richness and should be compatible with the theory of island biogeography.

However, the data from this study cannot be unequivocally explained by island biogeography theory. Therefore, the snapshot assumption may not be relevant and age may be a confounding variable. The demolition records needed to determine the age of the lots in this study were used in this study due to the short time available for fieldwork and the decentralization of municipal records. Potential sources of management variables (e.g. mowing, pesticide use, gardening) included records of complaints registered by Baltimore City's 3-1-1 telephone service and informal personal conversations with local residents. Future studies hopefully will have the ability to go through an interview process with local residents.

Although previous analysis of the vegetation architecture of vacant lots have shown them to fit within the theory of island biogeography (Crowe 1979), this study suggests otherwise. Neither species richness nor Shannon Weiner diversity indices supported the two predictions of the equilibrium theory of island biogeography in a Baltimore neighborhood. However, in light of the assumptions about age, management and microhabitat, the failure of island biogeography should not be surprising. In fact the failure of its assumptions is a stepping stone to understanding the many factors that can determine the phytogeography of urban vacant lots. It may be that other factors can play such a large role that the island biogeography characteristics are not a significant measure of variability. Another possibility is that the lots in Harlem Park are not isolated at all, and should be studied as connected corridors of vegetative habitat. There are many unmeasured variables that were present in this study, e.g. management and age. Thus, in order to effectively study immigration and extinction equilibriums in urban ecosystems, it would be beneficial to have a more controlled experiment, such as long-term observations with management variables. Future studies would benefit from doing a long term study on weed colonization patterns by tracking down management records from the City, analyzing the differences in soil and microclimate (Godefroid and others 2007) and determining the extent of the "rescue effect." Depending on rescue propagules of origins, this could determine whether high immigration rates are diminishing the extinction rates due to demographic and genetic contributions of immigrants. These contributions from immigrant species can increase the size and fitness of insular populations, thereby decreasing the probability that they will become extinct (Brown and Kodric-Brown 1977). This stabilizes populations, making them less sensitive to area and isolation as determining factors of immigration and extinction rates.

In the three decades after its creation, MacArthur and Wilson's eloquently simple conceptual Island biogeography theory has been the dominant influence on theoretical and empirical biogeography; however, over the last two decades that influence has begun to wane (Brown and Lomolino 2000). In empirical research, including this study, three key assumptions to the theory have been challenged:

1. In its original form the theory assumes that the independent and opposing forces of immigration and extinction rates affect on another, with extinction affected only by island area and immigration affected only by island isolation. This has been challenged by the occurrence of the "rescue effect" which has been a major influence to island biogeography and metapopulation dynamics (Brown and Kodric-Brown, 1977). Since the turn-over rate between immigration and extinction in the Baltimore lots was not measured, this phenomenon should be considered.
2. The second assumption supposes that species are essentially identical and islands are essentially identical except for critical differences between area and isolation. A number of studies have shown that differences between species and islands really do matter and can affect the assembly of insular biota by influencing the capacity to disperse across different kinds of barriers (Brown and Lomolino, 1989; Fox and Fox, 2000). Differences beyond area and isolation in the Baltimore lots, such as human transport, human management and lot heterogeneity, most likely exist and thereby confound the effects of area and isolation.
3. The third assumption to MacArthur and Wilson's theory is that the immigration and extinction rates are high enough such that patterns of species richness and composition result from a dynamic equilibrium between those opposing forces alone. This disregards nonequilibrial processes such as episodic extinctions associated with storms and tectonic activity (Whittaker and others 1989; Whittaker, 1995; Ward and Thornton, 2000). In the case of the Baltimore lots, sporadic weeding or other human activities could be comparable to these natural episodic extinctions.

These assumptions have led to developments of the theory of island biogeography, which include the alternative theoretical framework of a species-based theory of island biogeography (Brown and Lomolino 2000). This theory focuses on the resulting differences among both species and the island habitats in determining the assembly of insular biotas, and the resulting species richness and composition. By doing so, it preserves some of the pioneering insights from MacArthur and Wilson, particularly the observation that insular habitats reflect the interplay between recurrent immigration and extinction events, but not necessarily equilibrium between those two processes (Whittaker, 2000; Lomolino, 2000~~)~~.

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LITERATURE CITED

Audrey M., Porcher E., Devictor V., Arnal G., Moret J., Wright S., Machon N. 2008. Evaluation of floristic diversity in urban areas as a basis for habitat management. Applied Vegetation Science 11: 451-460.

Addinsoft. 1995-2009. XLSTAT Version 2010.2.01.

Baltimore City Department of Planning. 1960. Renewal Plan: Harlem Park Project II

Burch, W. R. J., Grove, J.M. 1993. People, trees and participation in the urban frontier. Unasylva 44:19-27.

Brown J.H. and Lomolino M.V. 2000. Concluding remarks: Historical perspective and the future of island biogeography theory. Global Ecology and Biogeography 9: 87-92.

Brown, J.H. & Lomolino, M.V. (1989) On the nature of scientific revolutions: independent discovery of the equilibrium theory of island biogeography. Ecology 70: 1954-1957.

Brown J.H. and Kodric-Brown A. 1977. Rates of insular biogeography: Effect of immigration on extinction. Ecology 58: 445-449.

Crowe, T. M. 1979. Lots of weeds: insular phytogeography of vacant urban lots. Journal of Biogeography 6:169-181.

City-Data.com. 2008. Harlem Park neighborhood in Baltimore, Maryland (MD), 21223, 21217 detailed profile.

City of Baltimore. 2004. ISTAR. [GIS file] ½ foot pixel resolution.

City of Baltimore. 2006. Feasibility Study for a Baltimore National Heritage Area. Retrieved from <http://www.ci.baltimore.md.us/government/heritage/downloads/102207%20NHA%20Feasibility%20Study.pdf.>

City of Baltimore. 2007. Baltimore City Parcel Boundaries.

DiChiro, G. 2002. Sustaining the 'Urban Forest' and Creating Landscape of Hope, in The Environmental Justice Reader, Adamson, J., Evans, M.M., and R. Stein, Editors. 2002, The University of Arizona Press: Tuscon, AZ. p. 234-307.

Fox, B.J. & Fox, M.D. 2000. Factors determining mammal species richness on habitat islands and isolates: habitat diversity, disturbance species interactions and guild assembly rules. Global Ecology & Biogeography 9: 19-37.

Gibbons, E. 1971. Stalking the wild asparagus. D. McKay Co., New York.

Gilbert, F. S. 1980. The equilibrium theory of island biogeography: fact or fiction? Journal of Biogeography 7:209-235

Gleason, H. A., Cronquist A. 1991. Manual of Vascular Plants of Northeastern United States and Adjacent Canada 2nd edition. New York Botanical Garden Press, Bronx, NY.

Godefroid S, Monbaliu D, Koedam N. 2007. The role of soil and microclimatic variables in the distribution patterns of urban wasteland flora in Brussels, Belgium. Landscape and Urban Planning 80: 45–55.

Graziano, P.T., Bainum, M.C., Dawson, K. 2007.Baltimore Housing: A Plan to Create the Baltimore City Land Bank. Housing Authority of Baltimore City- Department of Housing and Community Development. Baltimore, MD.

Haigh, M.J. 1980. Ruderal communities in English cities. Urban Ecology 4: 329-338.

Hodgkison S, Hero JM, Warnken J (2007) The efficacy of small-scale conservation efforts, as assessed on Australian golf courses. Biological Conservation 135(4):576–586

Holmgren, N. 1998. Illustrated Companion to Gleason and Cronquist's Manual: Illustrations of the Vascular Plants of Northeastern United States and Adjacent Canada. The New York Botanical Society Press, Bronx, NY.

JMP, Version 8. SAS Institute Inc., Cary, NC, 1989-2009.

Jules ES, Shahani P. 2003. A broader ecological context to habitat fragmentation: why matrix habitat is more important that we thought. Journal of Vegetation Science 14: 459–464

Kent M., Stevens R.A., Zhang L. 1999. Urban plant ecology patterns and processes: A case study of the flora of the city of Plymouth, Devon, UK. Journal of Biogeography 26: 1281-1298.

Lomolino, M.V. 2000. A call for a new paradigm of island biogeography. Global Ecology & Biogeography 9: 1-6.

Mac Arthur, R.H. & Wilson, E.O. 1967. The Theory of Island Biogeography. Princeton University Press. Princeton, NJ.

McKinney, M.L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127: 247-260.

Pickett, S. T. A., Cadenasso, M.L., Grove, J.M., Nilon, C.H., Pouyat, R.V., Zipperer W.C., Costanza R. 2001. Urban ecological systems: Linking terrestrial ecological, physical and socioeconomic components of metropolitan areas. Annual Review of Ecology and Systematics 32:127-157.

Rogers, G.F. & Rowntree, R.A. 1988. Intensive surveys of structure and change in urban natural areas. Landscape and Urban Planning 15:59-78.

Rudd H, Vala J, Schaefer VH. 2002. The importance of backyard habitat in a comprehensive biodiversity conservation strategy: A connectivity analysis of urban green spaces. Restoration Ecology 10(2):368–375

SAS Institute, Inc. 2009. JMP® 8.0.1.

Stearns, F.W.1971. Urban botany: An essay on survival. University of Wisconsin Field Standard Bulletin 4:1-6.

Ter Braak, C. J. F. 1986. Canonical correspondence analysis: new eigenvalue technique for multivariate direct gradient analysis. Ecology 67: 1167-1179.

Troy, A.R., Grove, M.J., O'Neil-Dunn, J.P.M., Pickett, S.T.A., Cadenasso, M.L. 2007. Predicting opportunities for greening and patterns of vegetation on private lands. Environmental Management 40:394-412.

Wilson, E.O. & Simberloff, D.S. 1968. Experimental zoogeography of islands: defaunation and monitoring techniques. Ecology 50:267-277.

Werner P. 1999. Why biotope mapping in populated areas? Deinsea 5: 9-26.

APPENDIX

FIGURE 1. A nine block portion of Harlem Park neighborhood in Baltimore, MD, showing the 31 vacant lots surveyed amidst the property parcels. The total area was 9 km².

TABLE 1. Quantitative data used for this study and their acronyms.

SPEC	SWDI	AREA	DISN	AW	DISL	L/W
Species richness	Shannon-Wiener Diversity Index	Area (m²)	Distance to the nearest lot (m)	Area of other vacant lots within 0.5-block buffer (m²)	Distance to the closest vacant lot (m)	Ratio of length divided by width

TABLE 2. Untransformed data for each lot. These variables (defined in Table 1) were used in regression analysis, principal component analysis and canonical correspondence analysis.

Lot Number	SPEC	SWDI	AREA (m²)	DISN (m)	AW (m²)	L/W
1	22	1.13	123.66	6.06	1702.78	6.33
2	18	0.95	286.76	3.33	2202.21	2.64
3	16	1.18	426.67	34.67	1187.94	1.70
4	22	1.06	476.12	3.33	1496.16	1.60
5	11	1.28	455.62	21.2	1501.34	2.08
6	18	1.25	132.56	15.23	304.71	5.78
7	17	1.31	167.13	15.23	2213.24	5.32
8	25	1.07	729.35	5.81	1851.86	1.61
9	16	1.50	461.81	10.69	2213.24	1.63
10	21	1.64	74.74	16.64	2363.00	6.67
11	19	1.13	130.57	9.69	2645.31	5.69
12	21	0.86	131.22	5.81	2363.00	5.68
13	35	1.50	479.77	0	3455.37	1.67
14	15	1.08	344.79	15.92	4734.09	3.55
15	25	1.63	141.94	0	4503.93	5.27
16	22	1.31	433.01	0	2070.08	2.08
17	13	1.09	455.27	23.94	2151.60	1.98
18	11	1.34	299.32	4.79	3227.34	2.77
19	35	1.23	411.41	2.4	3115.26	1.97
20	21	1.82	181.69	2.4	3067.06	5.20
21	14	1.44	559.29	0	2747.19	2.17
22	25	1.36	158.41	42.75	578.69	6.71
23	16	1.36	448.57	20.08	1302.17	1.94
24	31	1.12	147.16	31.7	580.88	4.85
25	31	1.56	154.21	22.9	813.75	5.28
26	18	1.25	673.52	21.83	1936.55	1.48
27	16	1.55	118.09	8.04	2252.07	5.39
28	13	0.94	107.99	3.07	2262.17	5.95
29	13	1.08	314.98	3.07	2055.18	2.58
30	12	1.46	699.95	21.7	3073.5	2.94
31	13	1.64	239.92	20.11	1276.3	1.96

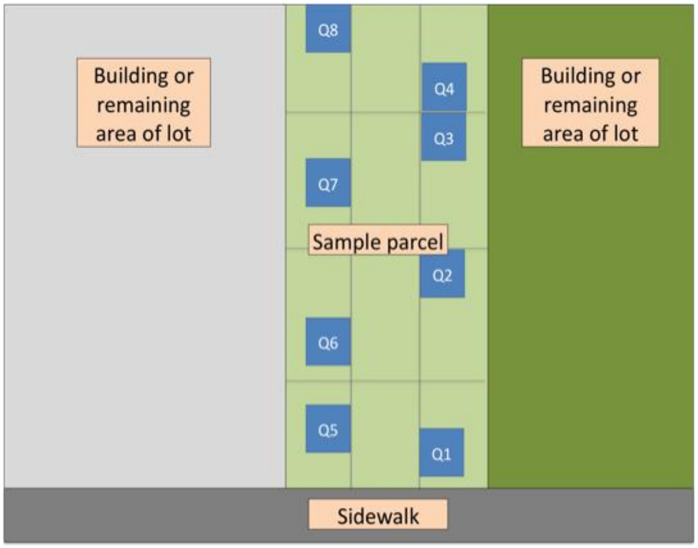

FIGURE 2. Sample design showing random placement of sample quadrats along transect lines with 5 m strata along the length of the parcel in a lot chosen to sample. Quadrats were labeled Q1-Q8 beginning on the street side and continuing down transects, which were labeled 1 and 2 beginning at the right. The building footprint was defined by Q1-3 and Q5-7, while relic backyards were defined by Q4 and Q8.

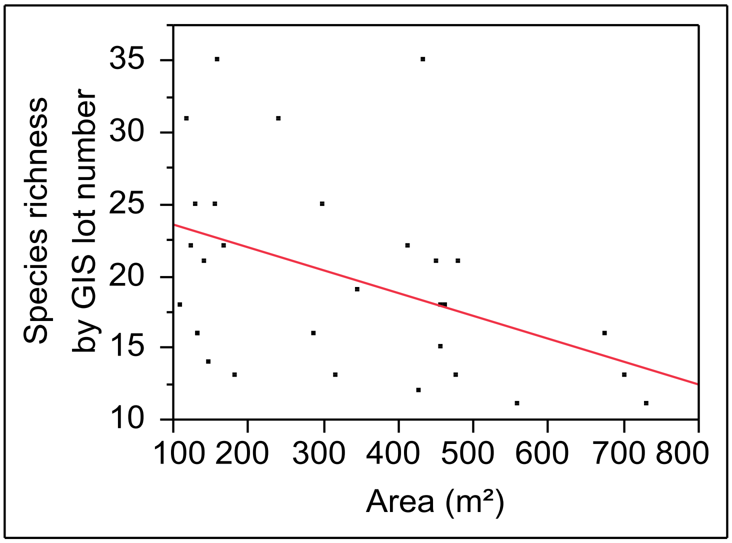

FIGURE 3. Linear fit of species richness within lot by lot area showing a significant negative regression characterized by the equation SPECIES RICHNESS = 23.688237 - 0.0129782AREA (R^2= 0.138, *p*= 0.019).

TABLE 3. Species unique to microhabitats within the lots are shown below. Front included the three quadrats along each transect closest to the street, corresponding to the footprint of the demolished building. Back included the quadrats furthest back from the street, corresponding to the former back yard of the demolished house.

SPECIES EXCLUSIVE TO FRONT
Ageratina altissima (L.) King & H. Rob.
Amaranthus retroflexus L.
Bromus erectus Huds.
Bromus sp.
Bryophyte 1
Bryophyte 2
cf. Malus sp.
Calystegia sepium (L.) R. Brown
Catalpa sp.
Humulus japonicus Siebold & Zucc.
Hypochoeris radicata L.
Lolium perenne L.
Potentilla simplex Michaux
Setaria pumila (Poiret) Roem. & Schult.
Silene latifolia Poiret
Solanum sp.
9 unique unknowns
Vicia sativa L.

SPECIES EXCLUSIVE TO BACK
Acer saccharum Marshall
Asclepias syriaca L.
Celastrus orbiculatus Thunb.
Celtis occidentalis L.
Linaria vulgaris Miller
Matricaria discoidea DC.
Polygonum perfoliatum L.
Robinia pseudoacacia L.
6 unique unknowns
Vitis sp.

TABLE 4. Summary of the results of regression analysis. AREA=area of lot, DISN= distance to the nearest lot, L/W= length over width ratio, AW= area of vacant lots within a 0.5-block buffer. R^2 signifies coefficient of determination, simple regression analysis, *R.S.S.* signifies the residual sum of squares and Prob > F signifies the probability greater than the Fisher value (SAS Institute, Inc. 2009). A 95% confidence interval was used to determine significance.

	SPECIES RICHNESS			SHANNON WIENER SPECIES DIVERSITY INDEX		
	R^2	*R.S.S.*	Prob > F	R^2	*R.S.S.*	Prob > F
AREA	0.193899	1309.429	0.0190	0.012193	1.244329	0.5759
DISN	0.006064	1309.429	0.6937	0.006267	1.244329	0.6888
DISL	0.0000478	1309.429	0.9721	0.003235	1.244329	0.7737
L/W	0.095958	1309.429	0.1087	0.004588	1.244329	0.7326
AW	0.005158	1309.429	0.7165	0.032073	1.244329	0.3618

TABLE 5. Linear relationships between environmental variables are represented by the regression coefficients below.

	F1	F2
SWDI	-0.290	0.666
Area	0.950	0.087
SPEC	-0.479	0.032
DISN	-0.223	-0.166
AW	0.081	-0.049
DISL	0.431	-0.559
L/W	0.112	-0.418

TABLE 6. Total basal area of tree species and their families found throughout sample area.

Family	Tree Species	Total basal area (m²)
Sapindaceae	*Acer negundo* L.	0.66
	Acer platanoides L.	0.07
	Acer rubra	0.25
	Acer saccharum Marshall	0.20
Bignoniaceae	*Catalpa* sp.	0.02
Cannabaceae	*Celtis occidentalis* L.	0.56
Fabaceae	*Mimosa* sp.	0.01
	Robinia pseudoacacia L.	0.49
Leguminosae	*Gleditsia triacanthos* L.	1.20
Moreceae	*Morus alba* L.	3.70
Rosaceae	c.f. *Malus* sp.	0.01
	Crateagus sp.	0.00
	Malus sp.	0.11
Paulowniaceae	*Paulownia tomentosa* (Thunb.) Siebold & Zucc. ex Steud.	3.49
Simaroubaceae	*Ailanthus altissima* (Miller) Swingle	1.35
Ulmaceae	*Ulmus parvifolia* Jacq.	0.44
	Ulmus sp.	1.80
	Zelkova serrata (Thunb.) Makino	0.14
Unknown	Unknown 15	0.01

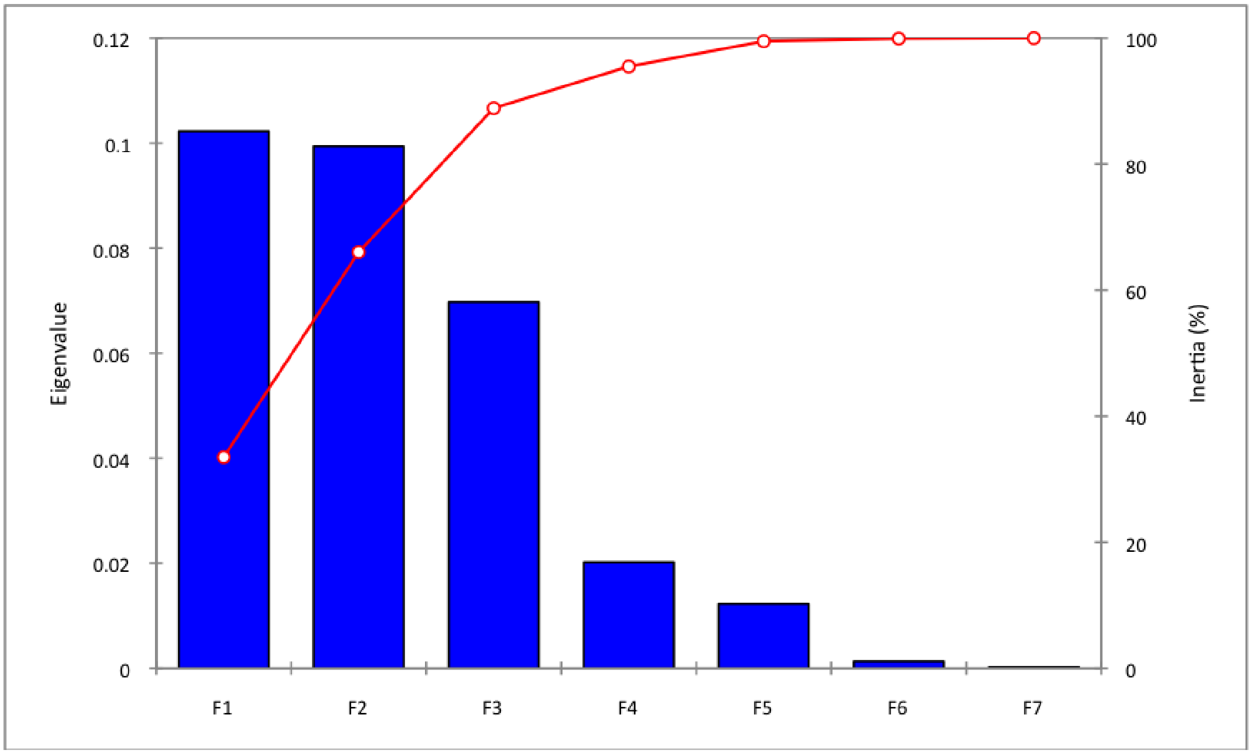

FIGURE 4. Scree plot showing the factors carrying the constrained inertia of correspondence analysis between the sites, environmental variables and the top nine species. Blue bars indicate Eigenvalues and red line indicates percent inertia. Eigenvalues F1 and F2 carry a total of 66.0% of the total inertia.

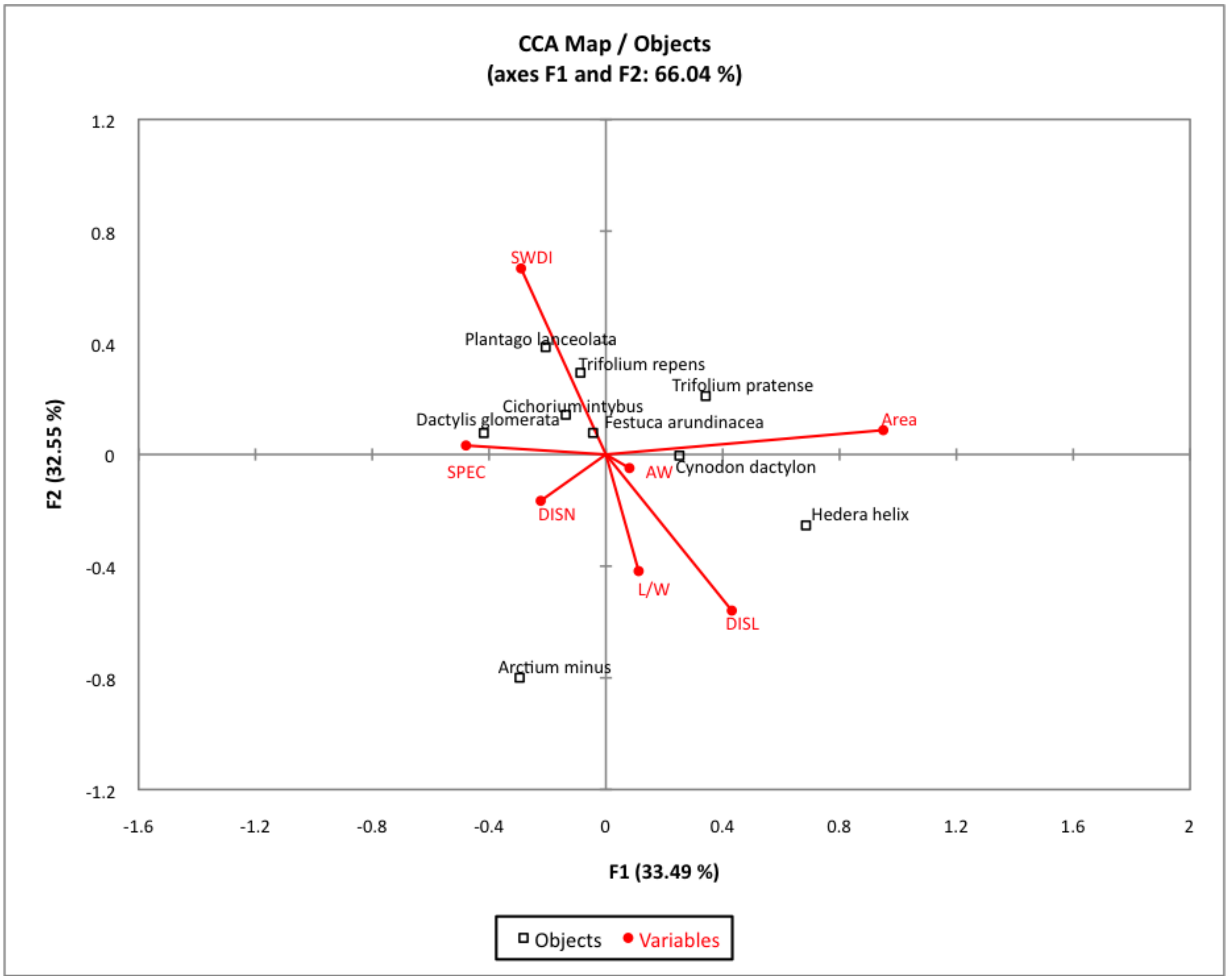

FIGURE 5. Ordination diagram of the first two axes of canonical correspondence analysis for the top nine plant species and eight habitat features used as quantitative variables. Arrows represent directions of greatest change in environmental variables. The nine most common species, representing 48.8% of all species collected, are in black. Environmental variables, in red, include AW (area of surrounding lots within half-block buffer), SPEC (overall species richness), AREA (m²), and SWDI (overall Shannon-Wiener Species Diversity Index).