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Carlisle R. Evans Peck
Augustana College - Rock Island

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“Over the River and Through the Woods”

An Analysis of Understory and Canopy Plant Diversity in Urban Riparian Forests

Carlisle Evans Peck

Augustana College, Rock Island IL

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Introduction:

Cities are rapidly becoming the dominant environment in which humans live. Currently, half of the world's population lives in urban centers, with the proportion projected to increase to two-thirds by 2050 (United Nations, 2012). As cities encroach on wild lands and as human interactions with nature increasingly take place in urban environments, it is vital to study if and how cities can function as ecosystems and how they can approximate the structure and function of the systems they replace. The norm in urban ecology has been to use a traditional ecological lens to study the biotic interactions in a city, and thereby determine how similar or different those interactions are. However, the contemporary literature exhibits a push toward a new urban ecology, one that recognizes the complex and interdisciplinary nature of urban systems. This would combine ecology with urban planning, socioeconomic studies, history, and anthropology, creating a new discipline, quite distinct from traditional ecology and tailored to understanding the unique systems that human proliferation has created (Steward et al 2008).

One of the most important facets of urban ecology to understand is biodiversity, not least because it is being lost at an unprecedented rate due to human actions (Chivian and Bernstein 2008) (Figure 1). It has been argued that biodiversity is critical for the stability and proper functioning of ecosystems, and therefore for the performance of crucial ecosystem services (Baskin 1994, Tilman et al 2006). Even rare species, sometimes thought to hold redundant roles, may support unique and significant functions in ecosystems (Mouillot et al 2013). Some studies suggest that it may not be species richness per se that contributes to ecosystem stability and function, but rather functional diversity (correlated, of course, with species richness) that is important in improving an ecosystem's ability to exploit available resources and protect against collapse as a result of stress (Baskin 1994). In any case, the diversity of species and interactions in an ecosystem are highly important for a functional ecosystem. Ecosystems perform important services for mankind and all life, some in the form of goods, others in the form of life-sustaining functions. Degradation and destruction of ecosystems have disrupted species interactions and ecosystem functions, and have required great expenditures of money and energy to "reinvent the wheel" and replace natural ecosystem services with anthropogenic methods (The Chicago Wilderness 1999, Committee on Noneconomic and Economic Value of Biodiversity 1999).

Vital services provided by ecosystems include harnessing and converting solar energy through photosynthesis, nutrient and waste cycling and retention, soil formation, climate moderation, flood control, water quality control, pollination, and pest control (The Chicago Wilderness 1999).Larsen, Turner, and Brooks (2012) simulated ecosystem services by creating computer models of global priority sites and random sites, and found that the former models performed better than the random models in terms of CO_2 sequestration and freshwater services (Larsen, Turner and Brooks 2012). A study by Pasari et al (2013) modeled the effects of local (α), intercommunity (β), and regional (γ) diversity on ecosystem functions. They found that individual functions were dependent on α diversity, while β and γ diversity affected complex interactions of multiple ecosystem functions. This suggests that diversity at multiple spatial scales, from local to landscape, is important to ecosystem functions. A connection between "human well-being" and biodiversity has also been proposed (Ulrich 1984, Chiesura 2004, Fuller et al 2007), but its details are less clear and there is some controversy over the nature of its effect, namely if perception of diversity correlates with actual diversity (Dallimer et al 2012). While it is common and natural to focus on human gain, the US EPA reminds us that "attempting to justify biodiversity only in terms of its utilitarian benefits to people will inevitably underestimate its true value" (The Chicago Wilderness 1999). Life has its own value independent of its utility, so we should be careful when justifying our concern for it only in terms of anthropocentric benefit.

As biodiversity is key to the proper functioning of ecosystems, protection and maintenance of biotic elements in expanding urban areas is crucial. Human alteration of the natural landscape is a major driver of biodiversity loss (Haines-Young 2009).While recent research suggests the potential of urban areas as functioning ecosystems, the general understanding is that there is no question that modern urban landscapes and the biota therein are drastically different from their pre-urban state (McKinney 2005). As McKinney (2005) points out, cities are systems that are maintained in a state of disequilibrium with natural processes solely at the grace of enormous inputs of energy. These inputs include irrigation in arid regions, drainage of wetlands, deforestation, reforestation in open areas, and imposition of impervious surfaces (McKinney 2005). It stands to reason, therefore, that the habitats found there will be something other than “natural”. For instance, although urban areas are sometimes more diverse and species-rich than rural areas, they exhibit very low beta-diversity, that is the species regimes among cities are very similar, often more homogeneous than among rural areas (Kuhn and Klotz 2006, Alvey 2006, Lundholm and Richardson 2010, McKinney 2005). A high percentage of these species are often exotics (Meek, Richardson, and Mucina 2010; Pauchard et al 2005; Shochat et al 2010, Kuhn and Klotz 2006, McKinney 2005). Those native species that do remain in urban environments tend to be limited to early successional species that are well adapted to the abiotic conditions and landscape structure that cities create (Pennington et al 2010, McKinney 2005). This global homogenization is not surprising given the fact that urban areas are very similar in structure, being built to accommodate the needs of a small subset of a single species (ie: the westernized and industrialized *Homo sapiens)* (McKinney 2005, Alvey 2006).

Furthermore, once altered, there’s no going back as abandoned lots and waste places will not revert to their natural state on their own, increasing the responsibility to devise ways to actively manage and protect urban ecosystems (Alvey 2006).

The question for the Urban Ecologist is: which aspects of urban structure cause the observed biodiversity patterns, and how can they be improved? Proposed causes include habitat fragmentation or loss (Pauchard et al 2005), patterns of human land use surrounding ecosystems (Meek et al 2010), socioeconomic status of surrounding neighborhoods (Kinzig et al 2005, Conway and Bourne 2013), history of nearby development (Conway and Bourne 2013), alteration of abiotic dynamics (Groffman et al 2003), and surrounding impervious surfaces (Farinha-Marques et al 2011, Pennington et al 2010).

Urban biodiversity patterns, however, may not be quite so simplistic as to be adequately characterized solely by homogenization. Taking the lens of conventional ecology has often led to the conclusion that urban centers are highly impaired ecosystems depauperate in plant and animal diversity, solely harboring homogenous communities, dominated by exotics, and situated among acres and acres of impervious hardscape. This result is seen as the consequence of the creation of wholly unfamiliar and novel habitats to which the vast majority of species are ill adapted (Lundholm and Richardson 2010). Such bleak conclusions often overlook the potential of urban areas to harbor unique biota and preclude the remediation of ecologically impoverished areas as an impossible feat. Yet, recent studies have begun to illuminate the importance of urban centers as ecosystems in their own right. For instance, urbanization often does exclude the presence of some species, but also invites others that are adapted to living in human environments (Shochat et al 2010). In fact, due to residential cultivation and the creation of greenspaces, urban species richness is often higher than in the surrounding countryside (Farinha-

Marques et al 2011, Kuhn and Klotz 2006, McKinney 2005) with a lower percentage of these being alien species (Meek et al 2010). Moreover, it has been suggested that even in highly altered locations, urban areas may provide habitats that are analogous to those found in the wild, and the species assemblages they support can be observed in similar wild habitats (Lundholm and Richardson 2010). This is a hopeful outlook that renews confidence in the ability of urban centers to be active ecosystems, and points to ways in which infrastructure can be altered to improve this function. Furthermore, it emphasizes that urban ecosystems are not merely disturbed and ruined habitats, but rather they are home to valuable species and are worthy of research, understanding, stewardship, and consideration (Crane and Kinzig 2005, Farinha-Marques et al 2011).

Riparian ecosystems are of particular interest to urban ecology because of their importance in mitigating many of the adverse effects of urban activity. They perform many valuable ecosystem services including “riverbank stabilization, nutrient cycling, flood attenuation, regulation of stream flows and stream temperatures, groundwater recharge, and water purification” (Meek et al 2009). Water quality is the main issue here. Riparian ecosystems act as “buffers” between upland regions and streams (Groffman et al 2003) Functional riparian ecosystems have the capacity to slow the transport of environmental pollutants from cities, namely NO_3^- (Groffman et al 2003). Unfortunately, human activity has altered the hydrology of many urban watersheds, rendering them largely impaired. This includes conversion of natural vegetation and soils to impervious hardscape or open parkland, channelization, or impoundment. The consequences of this hydrologic alteration are various. Channelization and impervious surfaces can cause stream systems to become sediment starved and “flashy”, leading to extensive erosion during rain events (Pickett et al 2008). This in turn affects the riparian ecosystem by reducing groundwater levels. Over time, this “hydrologic drought” has been shown to alter riparian plant communities as well as soil microbe communities. Groffman et al (2002) found that urbanization of riparian systems causes them to change from NO_3^- sinks to NO_3^- sources. Urban streams often have heightened levels of nitrogen and phosphorus (Pickett et al 2008). This altered hydrology also may provide the opportunity for colonization by invasive species by disturbing the established niches (Richardson et al 2007 in Meek et al 2009), which in turn could exacerbate further disturbance of ecosystem services (Tickner et al 2001 in Meek et al 2009).

This study focused on a group of forests within small urban watersheds in Rock Island County, Illinois. While many forested ravines are on private residential land, they are largely ignored and given little consideration. These are sites which, besides logging, have never been developed for human use, and are therefore more characteristic of the original natural landscape than much of the green space in the urban Quad Cities. Such sites hold the most promise for harboring native biodiversity and providing vital ecosystem services, making them of particular interest to protect (Breuste et al 2013). Our primary goal was to evaluate the current diversity and quality of the herbaceous and tree communities in these urban forests to illuminate the state of the urban ecosystem and evaluate its potential to integrate more native species. Our secondary goal was to identify and understand the anthropogenic factors influencing the observed diversity, including surrounding intensity of development, size of forest stand, and impervious surface. This knowledge is pertinent to developing plans to better manage and preserve these ecosystems for the health of the watersheds in question and the benefit of public well-being and the ecosystem at large. By studying these ecosystems under an urban ecological framework, taking into account both biological and anthropogenic influences, we hope to reach a detailed understanding of the Quad Cities urban ecosystem and provide more beneficial suggestions for moving forward.

Methods:

The Place: This study focuses on Rock Island County, IL. The cities of Rock Island and Moline, part of the Quad Cities metropolitan area, lie in Rock Island county. The county is also the site of the confluence of the Rock River and the Mississippi, with small tributary streams running through the two cities into both rivers. The urban forest sites were located in the cities of Rock Island (2010 pop: 39,018) and Moline (2010 pop: 43,977). This forest study was part of a larger project by the Upper Mississippi Studies Center researching the ecology of fifteen small riparian systems in the urban areas of Rock Island County. Each of the riparian systems studied were characterized by three features: 1) a headwater section in residential or low-density commercial areas, 2) a forested ravine characterized by steep slopes resulting from the stream's incision of the river bluffs, and 3) a channelized or underground confluence section on either the Rock or Mississippi River floodplain flowing through high density commercial land use and impervious surfaces. This study surveyed the biodiversity of the herbaceous and tree communities in the ravines section of 8 of the studied riparian systems. Three ravines were situated in Rock Island, four in Moline, and one was a rural reference site in Loud Thunder Forest Preserve outside of Andalusia, IL.

Sites 1 and 22 were both selected based on their location in forest preserves, the former in the urban Blackhawk State Historic Site in Rock Island, and the latter in the rural Loud Thunder Forest Preserve in Andalusia. The remaining 6 sites were each selected based on the land use surrounding them. Using ArcGIS, a point was selected at roughly the center of each forest stand,

and the percentage of each of the following NLCD land use classification was determined: Open Ground (21), Forest (41), and Low (22), Medium (23), and High density (24) urban. We chose as great of a spread of land use regimes as possible, from more developed (Site 13, 8.6% forest) to more natural (Site 7, 32.15% forest). As these sites are all on private property, in the backyards of many residences, access was also taken into consideration when making the choice.

Sampling: Three sampling methods were used in each of the eight study sites. First, the line-point survey method was used to sample the herbaceous understory community. At each site, four 50m transects were laid out, two on each side of the stream. There are a few exceptions to this guideline: at site 4 an additional 25m transect was sampled, at site 7 access issues only permitted the sampling of one side of the stream with 2 transects, at site 10 one side of the stream did not have sufficient vegetative cover to permit sampling, and at 13 only three transects were sampled. The sampler would walk along the transect, stopping every 0.50 m and dropping a roughly 2m tall flag. At each point, the identity of each species touching the stake was recorded from the top down. If one species contacted the point multiple times, it was still only recorded once. The ground cover was also recorded at each point as leaf litter, moss, sticks, bark, branch, or bare soil. During herbaceous sampling, any species identified other than those caught by the line-point sampling were noted.

The trees at each site were sampled using the point-centered-quarter plotless technique. This method aimed to gather ten points of data on each side of the stream, for a total of 20 points of data per ravine site. This was not possible at site 10 as one side of the stream is not forested. At each point, a set of metal cross-bars was laid out in no specific orientation, dividing the forest into four quadrants. In each quadrant, the identity of the nearest woody stem (including trees, shrubs, and vines) measuring at least 1 inch (~2.5 cm) in diameter was recorded along with the distance to the base of that stem and its diameter at breast height (DBH). The next point was determined by pacing off 15 steps (app. 15m) parallel to the stream.

In the process of herbaceous sampling, many plants were left unidentified as they had either senesced or had not matured enough. To help include the latter group, those species that had not yet bloomed over the summer, each site was visited again in mid-September and haphazard sampling was conducted for 20 minutes, identifying any novel individuals.

GIS Work: Eight environmental predictor variables for forest overstory and understory biodiversity were calculated using ArcGIS 10.1: mean imperviousness of surfaces within a 250m and a 500m radius, percent developed cover within 250 m and 500m radius (NLCD 2007 22, 23, 24), percent open space within 250 m and 500 m radius (NLCD 2007 21), and percent forest cover within 250 m and 500 m radius (NLCD 2007 41). Mean imperviousness and percent developed cover were intended as proxies for urbanization intensity, and percent forest cover is a proxy for habitat patch size. Linear regressions were performed in Microsoft Excel 2007 with these environmental predictor variables and the FQI, Species Richness, understory invasive cover, overstory invasive density, Shannon's diversity index, for each site.

Analysis:

Understory Herbaceous Community: Line-intercept data was used to calculate absolute and relative cover for each species per site. Non-metric multidimensional scaling (NMS) ordination was used to relate patterns in community composition to environmental gradients (PC-Ord™; McCune and Grace 2002). Joint plots and Pierson's correlations were used to describe relationships between environmental gradients and the strongest patterns of community composition.

Hierarchical agglomerative cluster analysis was used to identify groups of sites differing in community composition (McCune and Grace 2002). Multi-variate differences in community composition between identified groups were tested using multi-response permutation procedures (MRPP) ($\alpha = 0.05$). Identified groups were overlaid onto ordinations to accentuate relationships between groups and environmental gradients. Multivariate differences in environmental variables between groups were tested with MRPP. Differences in individual environmental variables between groups were assessed with ANOVA ($\alpha = .10$), and Bonferroni-adjusted 90% confidence intervals were used to quantify differences between groups.

Indicator Species Analysis (ISA) was used to quantify community composition and determine the herbaceous species uniquely associated with the groups of sites identified by cluster analysis.

A separate FQI value was calculated for the herbaceous understory community as well.

<u>Overstory Tree Community:</u> Using the point-quarter data, the total average tree density was calculated for each transect by averaging point-to-stem distances, squaring that value, and then taking the inverse. The average density of each species was calculated by dividing the frequency of that species by the total number of trees sampled in a transect (40) and multiplying by the total tree density. These values for each transect in a site were averaged, and an ordination and indicator species analysis was performed on this density data.

Non-metric multidimensional scaling (NMS) ordination was used to relate patterns in overstory community composition to environmental gradients (PC-Ord™; McCune and Grace 2002). Joint plots and Pierson’s correlations were used to describe relationships between environmental gradients and the strongest patterns of community composition.

Hierarchical agglomerative cluster analysis was used to identify groups of sites differing in community composition (McCune and Grace 2002). Multi-variate differences in community composition between identified groups were tested using multi-response permutation procedures (MRPP) ($\alpha = 0.05$). Identified groups were overlaid onto ordinations to accentuate relationships between groups and environmental gradients. Multivariate differences in environmental variables between groups were tested with MRPP. Differences in individual environmental variables between groups were assessed with ANOVA ($\alpha = .10$), and Bonferroni-adjusted 90% confidence intervals were used to quantify differences between groups.

Indicator Species Analysis (ISA) was used to quantify community composition and determine the tree species uniquely associated with the groups of sites identified by cluster analysis.

Overall**:** All of the species observed at each site were compiled in a master list, giving the species richness of each site. Each species was then assigned a coefficient of conservatism obtained from Taft et al (1997). A Floristic Quality Index value was derived for each site using the following equation:

$$I = (\sum C \times \sqrt{N})/N$$

Where C is the coefficient of conservatism and N is the number of taxa.

Results:

Overall: Combining the species found in the herbaceous, tree, and haphazard sampling methods, Site 1 was found to have the greatest species richness (98). The lowest species richness (43) was found at site 7. These sites also had the highest and lowest FQI's, respectively: 24.34 and 14.03. The highest invasive percentage of the total species observed was found at site 4

(33.8%), and the lowest at site 10 (18.5%). The highest relative cover of invasive species was found at site 13 (34.4%) and the lowest was found at site 22 (8.8%).

Regression analysis found overall FQI to be positively correlated with species richness (r^2=0.76) and understory species richness (r^2=0.5264). Further linear regression analyses revealed that FQI was negatively correlated with mean imperviousness of surfaces within a 250m radius (r^2=0.429). Species richness was positively correlated with percent forest (r^2=0.639) and negatively correlated with mean imperviousness (r^2=0.7689) and percent development (r^2= 0.6117).

Understory Herbaceous Community: Site 2 was revealed to have the highest understory species richness (56), while 7 had the lowest (21). The highest Understory FQI was found at Site 6 (19.59), and the lowest at site 7 (9.82). Site 13 had the highest invasive cover (34.37%) while Site 22 had the lowest (8.834%). The only predictor variable that correlated with the understory data was Percent Open Space within a 250m radius, which was correlated negatively with understory species richness (r^2=0.4141) and understory Shannon's Diversity Index (r^2=.7042) (Panel 1).

Nearly 91% of variation in herbaceous community composition was explained by the final ordination (Fig. 3). Axis 1 was the dominant axis explaining 75.2% of variation in composition data. Axis 1 was a strong gradient of decreasing impervious surface (r = - 0.72), decreasing levels of all three densities of urban development, and increasing forest cover (r = 0.79) (Fig. 3). Axis 2 represented a much weaker relationship explaining 15.6% of the remaining variation in herbaceous species cover. The amount of high density urban development decreased (r -0.42) along this axis. The strong gradient of decreasing urbanization/increasing forest cover had positive correlations with *Amphicarpa bracteata, Ulmus rubra*, *Cryptotaenia canadense*, and *Cornus racemosa* and negative correlations with *Circaea quadrisulcata, Parthenocissus quinquefolia, Acer negundo*, *Eupatorium rugosum* and *Alliaria officinalis*.

Cluster analysis identified three distinct groups of understory herbeacous communities with 0% of the information remaining (MRPP using species data: A = 0.21, *P* < 0.015). The ordination of the herbaceous understory data revealed that the eight sites split cleanly into three groups: 1.) Sites 1, 2, and 13; 2.) Sites 4, 6, 7, and 10; and 3.) Site 22. Groups 1 and 2, which were comprised of the urban forests, were more alike each other than either were to Group 3, the rural reference site. The results of an indicator species analysis revealed the species that are most closely associated with Groups 1 and 2 (this analysis could not be run on group 3 as it only contains one species). The indicator species in Group 1 include *Acer saccharum* (IV=89.9, p=0.0844), *Celastrus scandens* (IV=82.4, p=0.0492), *Duchesnea indica* (IV=100, p=0.0244), *Eupatorium maculatum* (IV=92.4, p=0.077), and *Ligustrum vulgare* (IV=92.8, p=0.0492). The indicator species in Group 2 include *Acer negundo, Alliaria officinalis* (IV=64.1, p=0.0818), and *Circaea quadrisulcata* (IV=77.7, p=0.0244). Since the indicator species for Group 3 could not be determined, the five species with the highest percent cover were chosen for comparison: *Apios americana* (4.73%), *Laportea canadensis* (32.88%), *Leersia* spp. (4.5%), *Sambucus canadensis* (6.3%), and *Sanicula marilandica* (6.1%). The intensity and density of urbanization and forest patch size (amount of habitat fragmentation) also differed significantly among groups (MRPP using environmental predictor data: A = 0.07, *P* < 0.011; Fig. 3). A t-test revealed that groups 1 and 2 differed significantly in invasive cover (p=0.063) and percent forest (p=0.0516) (Panel 2).

Overstory Tree Community**:** The tree sampling revealed that Site 1 had the highest species richness (19) while site 10 had the lowest (10). Tree community FQI was highest at Site 22 (12.73) and lowest at Site 6 (9.25). Linear regression analyses revealed that tree community FQI was positively correlated with % Forest cover within a 250m radius (r^2=.5654) and negatively correlated with mean imperviousness of surfaces within a 250 m radius (r^2=.4658) (Panel 1). The correlation was weaker for the variables within a 500 m radius. Interestingly, tree species richness was not correlated with these variables.

Nearly 81% of variation in tree overstory community composition was explained by the final ordination (Fig. 4). Axis 2 was the dominant axis explaining 66.8% of variation in composition data. Axis 1 represented a weaker relationship explaining 13.5% of the remaining variation in tree species cover. Axis 1 was a strong gradient of increasing impervious surface (r = 0.78), increasing medium density urban development (r = 0.71), and decreasing forest cover (r = -0.73) (Fig. 4). The species that were positively correlated with Axis 1's strong gradient of decreasing urbanization were Sycamore, Grey Dogwood, Cottonwood, and Hophornbeam, while those that were negatively correlated were Boxelder, Redbud, and Black Cherry.

Cluster analysis identified four distinct groups of overstory communities with 0% of the information remaining (MRPP using species data: A = 0.18, *P* < 0.005). Group 1 includes sites 1, 2 and 22; Group 2 sites 4 and 7; Group 3 sites 6 and 10; and Group 4 site 13. The results of an indicator species analysis revealed the species that are most closely associated with Groups 1-3 (this analysis could not be run on group 4 as it only contains one study site) (Table 1b). The results of an indicator species analysis revealed the species that are most closely associated with each of the groups. 1.) *Cornus racemosa* (IV=100, p=0.0276) and *Ostrya virginica* (IV=100, p=0.0276); 2.) *Prunus serotina* (IV=80, p=0.0346), *Acer negundo* (IV=70, p=0.1094), *Cercis* *canadensis* (IV=100, p=0.0874), *Ailanthus altissima* (IV=62, p=0.1067); 3.) *Celtis occidentalis* (IV=67, p=0.0254) and *Quercus alba* (IV=83, p=0.1076). As an ordination could not be performed on group 4 (Site 13), the three species with the highest average density were selected: *Prunus serotina* (0.0234/m^2), *Quercus velutina* (0.01456/m^2), and *Acer saccharum* (0.02526/m^2). Contrary to the herbaceous understory results, the intensity and density of urbanization and forest patch size (amount of habitat fragmentation) did not differ significantly among groups (MRPP using environmental predictor data: A = 0.05, $P < 0.190$). A one-way ANOVA showed that groups 1, 2, and 3 differed significantly in terms of species richness (F=4.6274, p=0.09107), FQI (F=8.778, p=0.0344), and percent forest cover within a 250 m radius (F=5.62, p=0.0688) (Panel 3).

Discussion:

Herman et al (2001) suggest that an FQI of greater than 35 is indicative of a “floristically important” site, while those of less than 20 are of “minimal significance” (Herman et al 2001). Using this judgment, none of the Quad Cities forests sampled are “floristically significant”, yet some lie between those two thresholds. It is rather shocking and disconcerting that, even within state-protected natural areas such as Blackhawk State Park (site 1) and Loud Thunder Forest Preserve (Site 22), an FQI of only 24.34 and 21.29, respectively, were found. Nonetheless, this suggests that there are a good amount of native species present, though they may largely be characteristic of early successional and recently disturbed habitats (citation). Sites 1, 6, 10, 13, and 22 all have FQI values between 20 and 25, suggesting that they are not only worth noting but also may be important sites to consider for future conservation efforts as they seem to contain a valuable framework for more native flora. Overall, our data support the contention that urban ecosystems can contain valuable biodiversity, though all of the sites would benefit from serious conservation efforts.

The results of the regression analyses show that the intensity of urbanization has a strong negative effect on the native plant community. The FQI of a site was correlated positively with herbaceous understory community species richness, and negatively with mean surface imperviousness within a 250m radius (Panel 1). The correlation with understory species richness suggests that the understory community may be the major indicator of the overall floristic quality of a site. Furthermore, the negative correlation with mean impervious surface suggests that the presence of urban development near a site has a negative effect on the plant community therein, a result consistent with the findings of Pennington et al (2010). Species richness was also negatively correlated with mean imperviousness, and positively correlated with percent forest cover, also consistent with Pennington et al (2010). The latter suggests that patch size is a significant indicator of species richness, which is consistent with the study by Pauchard et al (2005) on the effects of habitat fragmentation in urban areas. Regression analyses of the understory herbaceous community and overstory tree community data also support the negative effect of habitat fragmentation as tree community FQI was positively correlated with % Forest cover within a 250m radius, and urbanization as tree community FQI was negatively correlated with mean imperviousness of surfaces within a 250 m radius, while understory species richness and understory Shannon’s Diversity Index were both negatively correlated with percent open space within a 250m radius. The loss in species richness could be a result of altered hydrologic regimes, damaging behaviors of residents that accompany development, or edge effects resulting from the size of the forest plot. Indeed, for some sites the distance between the center of the forest and surrounding neighborhoods was very slight, namely site 7 which had the lowest percent forest cover, FQI, and species richness and the highest mean imperviousness This points to a possible restoration strategy: that is, removing impervious surface and development surrounding these forests and replacing it with as much natural vegetation as possible.

All of the urban sites (1, 2, 4, 6, 7, 10, and 13) were revealed to be more similar in understory community composition to each other than they are to the rural site (22). This result is consistent with studies on urban-rural gradients that have found urban habitats to be more like one another than they are like rural habitats (Kuhn and Klotz 2006, Alvey 2006, Lundholm and Richardson 2010, McKinney 2006). The second significant separation splits the urban sites into those in Rock Island (Group 1; 1, 2, and 13) and those in Moline (Group 2; 4, 6, 7, 10). Group 1 had significantly greater forest cover within a 250 m radius and a significantly higher average Shannon's Diversity index (Panel 2). This suggests that the size of the forest, as a proxy for habitat fragmentation, significantly affects understory species composition, in accordance with the findings of Pauchard et al (2005) on habitat fragmentation. As the Rock Island sites are all in parks (1 and 2 in Blackhawk State Historic Site, and 13 near McCandless Park) while the Moline sites are primarily backyard ravines (with the exception of 10 in Riverside Park), forest size seems to adequately explain the ordination results. The ordination also shows that certain species are strongly associated with a gradient of increasing urbanization. *Amphicarpa bracteata, Ulmus rubra, Cryptotaenia canadense*, and *Cornus racemosa* were positively correlated with percent forest within 250m at a site and negatively correlated with mean imperviousness within 250m. These species are all native, and are characteristic of more conservative habitats (*A. bracteata* C=4, *U. rubra* C=3, *C. canadense* C=1, *C. racemosa* C=2) Meanwhile, *Circaea quadrisulcata, Parthenocissus quinquefolia, Acer negundo, Eupatorium rugosum,* and *Alliaria officinalis* all showed the opposite relationship. This makes sense, as these species are either invasive (*A.*

officinalis) or disturbance tolerant (*C. quadrisulcata* C=2*, P. quinquefolia* C=2, *A. negundo* C=1, *E. rugosum* C=2) (Table 1a)

Explaining ordination of the tree density data is less clear. They split into four groups, Group 1 included the sites in protected areas (Blackhawk SHS and Loud Thunder) and Group 2 included sites that superficially were somewhat decrepit. The other groupings are less easy to explain than those based on the herbaceous data. Groups 1 and 3 were largely comparable, being characterized by significantly higher species richness and FQI than Group 2, and Group 1 had significantly higher percent forest cover (Panel 3). As all of the sites in Group 1 are in or near forest preserves, it is expected that forest cover should be significantly higher. Meanwhile, understory species regimes are not correlated with any of these variables aside from forest cover. This discrepancy could be due to “the ghost of development past”, that is, the fact that the consequences of any human actions would take longer to manifest in the tree community than they would in the herbaceous understory, therefore the presently observed tree communities may reflect conditions of a decade or more ago. The ordination also showed that certain overstory species are negatively associated with a gradient of increasing urbanization including Sycamore, Grey Dogwood, Cottonwood, and Hophornbeam, while Boxelder, Redbud, and Black Cherry were positively associated with urbanization. Boxelder and Black Cherry are disturbance tolerant species, and while Redbud is not its prevalence as an ornamental tree point to a possible mechanism behind its association with heavy urbanization. It should be noted, however, that this axis only explained 13% of the canopy ordination results. Axis 2 explained 66.8%, but no predictor variables were correlated with it. This could reflect variables we did not account for, namely “the ghost of development past”. That could also explain why it is that the overstory community seems to be independent of the understory community.

The results of the herbaceous understory indicator species analysis show the importance of invasive and early successional species. Especially disconcerting is that *Celastrus scandens* and *Ligustrum vulgare* are among the indicator species for Group 1 and *Alliaria officinalis* for Group 2. These are incredibly prolific invasive species, and their presence should be monitored closely. The presence of *Alliaria officinalis,* moreover, was likely underestimated since it is a spring blooming plant which mostly senesces by midsummer, when sampling was begun. While captured by neither understory nor canopy data, the choking presence of large *Rosa multiflora* and *Lonicera spp* shrubs, namely at the fringes of Blackhawk State Historic Site (Site 1), is cause for great concern due to the virulent nature of these exotics. The relative cover of invasive species in the understory was also quite high, at Site 13 exceeding 1/3 (34%).

While many studies of urban ecology use an urban to rural gradient to determine the ecological effects of urbanization, our study focuses on an urban gradient. This focus has allowed us to look more closely at the diversity of habitats within an urban environment, consistent with the urban ecological model, rather than simply portraying an “urban vs. rural” dichotomy. Although the urban sites were more alike than they were like the rural site in terms of understory herbaceous diversity, the variation among urban sites is enough to question the simplicity of the “urban vs. rural” split. This also shows the potential for β diversity, which can be important for the performance of certain ecosystem functions, as Pasari et al (2013) showed.

Further studies should attempt to expand the scope of this project in the Quad Cities. Our selection was restricted to the urban riparian systems the UMSC was studying, but there are many other forested ravines to be studied. It also may be useful to analyze the history of urban development surrounding these pockets of forest as well as the time since last overstory removal, as it has been suggested that age of development has a significant effect on the quality of an ecosystem (Conway and Bourne 2013). Soil composition also may have a significant effect on the forest communities. Further studies also should investigate the health of the ravines in higher resolution, using more transects within a forest to tease apart a finer scale of influences on diversity. Moreover, our results suggest the importance of an edge effect, and this phenomenon also should be investigated more closely. Directly investigating the relationship between stream water quality and forest diversity would also be an interesting and useful practice, as studies have shown the important link between plant communities and hydrologic variables. A direct study of the activities and attitudes of the landowners surrounding the ravines could more clearly elucidate further directions in improving the relationship between residential areas and natural places.

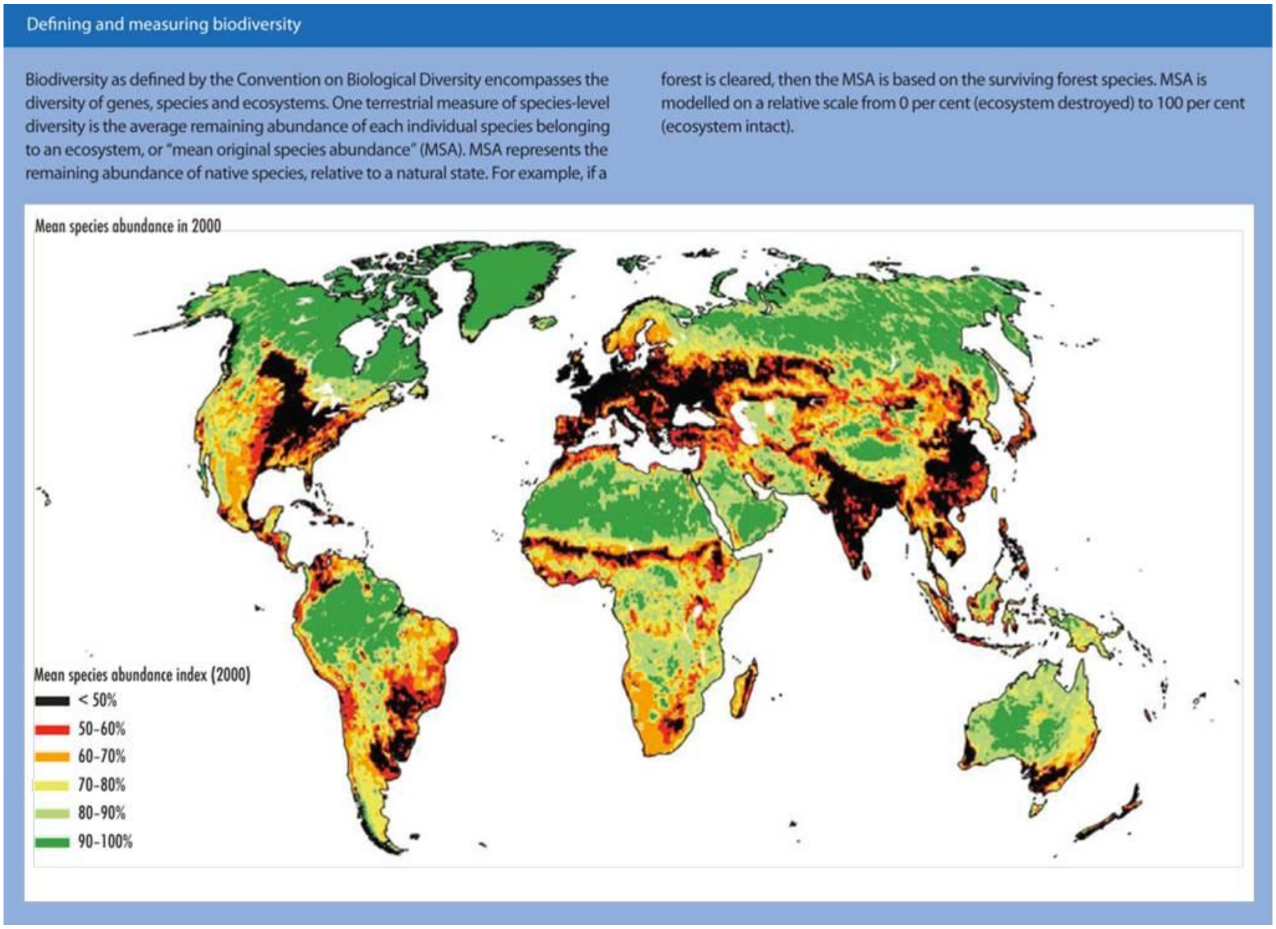
The GIS analysis of the surrounding land cover and impervious surface also has its downsides, namely in resolution. The NLCD classifications are based on 30X30 m pixels, a resolution which potentially misses a great deal of the actual landscape. Future studies could refine the land cover analysis so that it more accurately represents the real heterogeneity of the urban landscape. The Hercules method of land cover analysis show promise as a very fine-resolution method, and the Upper Mississippi Studies Center is currently looking into employing it in future iterations of its research program.

The salient question is, now that the state of these ravine ecosystems has been characterized, where do we go from here? Certainly, some of these forests harbor valuable native biodiversity, and their functionality as natural systems has the potential to improve with proper restoration and management. But what is the best method of going about this? It may be that the biggest remediation needs to take place in the minds of the citizenry. Miller (2005) laments "the extinction of experience in urban areas", referring to the estrangement of the public from the natural world that augments ecological apathy, which in turn enables and perpetuates environmental abuse and destruction. It results from the fact that “native species that remain in cities worldwide tend to be segregated from the neighborhoods where most human residents live” (Miller 2005). Yet our research has proven this claim to be false for the ravines we studied: they are immediately adjacent to residences, yet they are still ignored. This illustrates the necessity of an attitudinal shift on the part of the public, for although these forests are quite literally in the backyards of residences and are therefore physically salient parts of the immediate environment for many people, they are not a part of the immediate perceptual environment, and are largely ignored by their human neighbors as derelict waste places suitable for only for trash disposal. It is clearly not enough to conserve a habitat by leaving it undeveloped; rather it may be necessary to explicitly draw public awareness to these places as places of value. One common way of managing natural systems in an urban environment is by converting them into places of recreation such as nature preserves or parks (Hager et al 2013). This can be accomplished through the building of trails and shelters.

Remediation of urban waste places in the form of human recreation areas has the added benefit of countering the “extinction of experience”, Even when there is sufficient parkland and open space in an urban center, the flora and fauna with which the majority of people interact is highly altered or exotic, and many are notoriously inept at distinguishing between native and invasive species, which hampers progress on protecting native biodiversity (Dallimer et al 2012). The new paradigm of “reconciliation ecology” speaks to this issue, and urges the integration of the urban with the natural not by trying to approximate the original ecosystem that has been lost, but by modifying the urban structure to foster the growth and proliferation of native species (Miller 2005). As these urban forest sites are all right next door to residential areas, they provide a particularly good opportunity to demonstrate the importance of urban nature to residents. At the same time, there is evidence that unmanaged and more “wild” patches may be more functional and diverse than those that are managed (Farinha-Marques et al 2011). This could be due to the propensity of human presence to unintentionally disturb habitat and introduce invasive species.

Modifying the urban structure may require some changes to the residential areas surrounding the forested ravines. Since it would be rather rude to evict homeowners in those properties, it instead may be necessary to urge homeowners to treat their backyards as forests, rather than as traditional residential zones with perfectly manicured lawns. The latter aesthetic has become the norm, but it also has had profound consequences of environmental degradation (Robbins and Sharp 2003). This means a conversion from cultivated to native vegetation, reduction in open lawn area, and termination of chemical fertilizer and herbicide application. Robbins and Sharp in “The Moral Economy of the American Lawn” suggest that the suburban American Lawn is managed primarily for the “preservation of community, family, and environment” rather than physical resources and ecological integrity. This is because the values of stability and affluence have been connected with lawn monocultures, while unmanaged areas (ie: our forested ravines) are stigmatized as dereliction and waste. Clearly a full aesthetic and socioeconomic overhaul of suburban American values are necessary to reverse the tendency remove the natural and unmanaged from consideration. But they also suggest that meaningful change needs to begin at the level of local landowners, arguing that “local people perform as land managers and local agents...[and are] as much producers as they are consumers of landscapes” (Robbins and Sharp 2003).

Furthermore, the economic, ecological, and social benefits provided by functional urban natural systems are steep enough to heavily incentivize their restoration and maintenance (Hager et al 2013, Chiesura 2003, Larsen et al 2012). While our stream research results suggest that the ravines are performing significant services in protecting water quality, the fact that water quality overall is substandard indicates that the functionality of the riparian systems could be improved significantly (Reisner et al., unpublished). The Baltimore Ecosystem Study (Hager et al 2013) demonstrates that the ecological restoration of a watershed not only can improve water quality, but can also revitalize the surrounding neighborhood. This type of solution could greatly benefit the Quad Cities area. Though this research has certainly identified the urban ecology of the Quad Cities has some room for improvement, the most important message of this study is one of hope. While the forested ravines of Rock Island and Moline are not ideal examples of Midwestern upland hardwood forest, they nonetheless are not overrun by exotics, they contain valuable native diversity, and they hold the potential for remediation by the seamless integration of the human and the natural in an urban ecological framework.



Defining and measuring biodiversity

Biodiversity as defined by the Convention on Biological Diversity encompasses the diversity of genes, species and ecosystems. One terrestrial measure of species-level diversity is the average remaining abundance of each individual species belonging to an ecosystem, or "mean original species abundance" (MSA). MSA represents the remaining abundance of native species, relative to a natural state. For example, if a forest is cleared, then the MSA is based on the surviving forest species. MSA is modelled on a relative scale from 0 per cent (ecosystem destroyed) to 100 per cent (ecosystem intact).

Fig. 1: A world map showing the average species abundance remaining, relative to the natural, preindustrial state. Notice that densely populated urban areas are often the most depauperate (ie: Eastern North America, Europe, India, and Eastern China) (Citation).

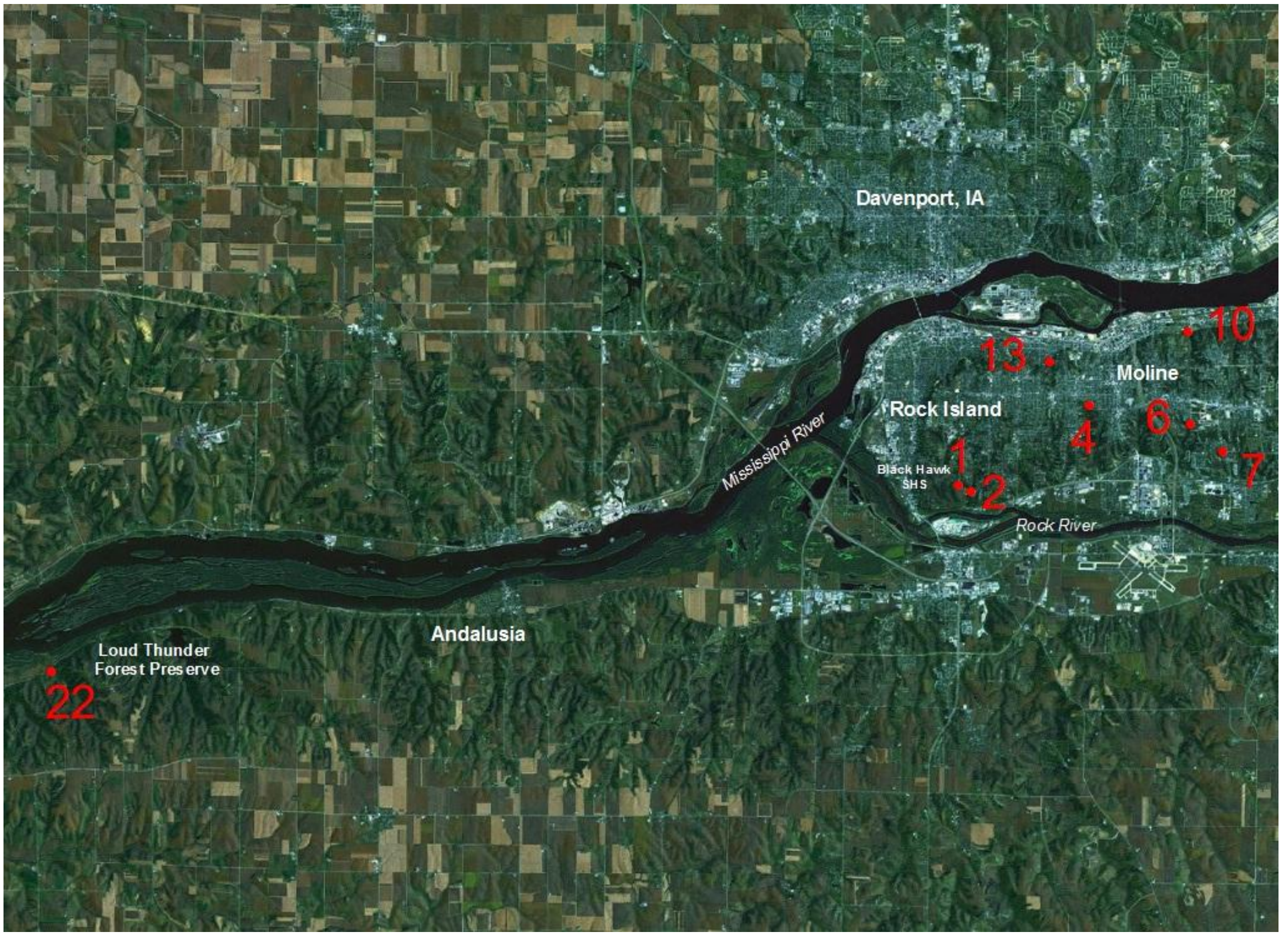


Fig. 2: A map of the Quad Cities area showing the location of each of the forested ravine sites.

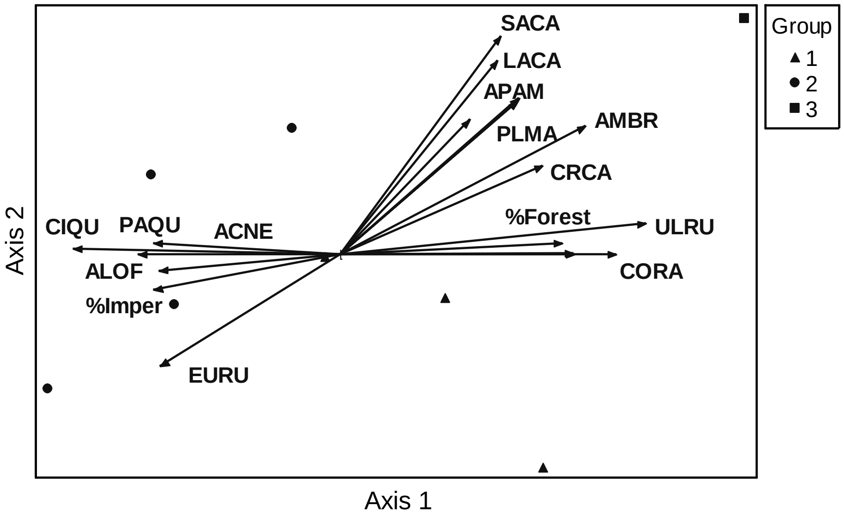


Figure 2: Ordination of study sites in herbaceous community composition space. NMS ordination with final stress of 7.67; final instability of < 0.01; Monte Carlo test P-value < 0.05. Vectors show the strength and direction of correlations between species cover and environmental predictor variables and axes. Only variables with significant R-squares (> 0.40) are shown. Different plot symbols show groups derived from cluster analysis that differ in composition and environmental factors.

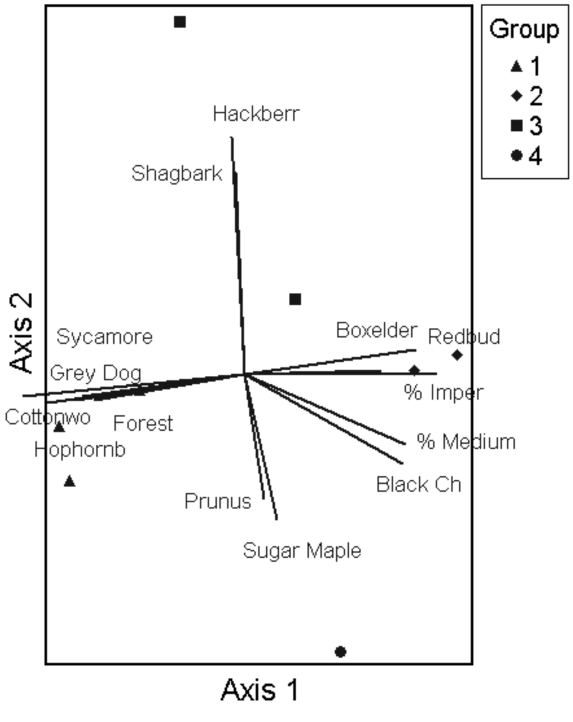
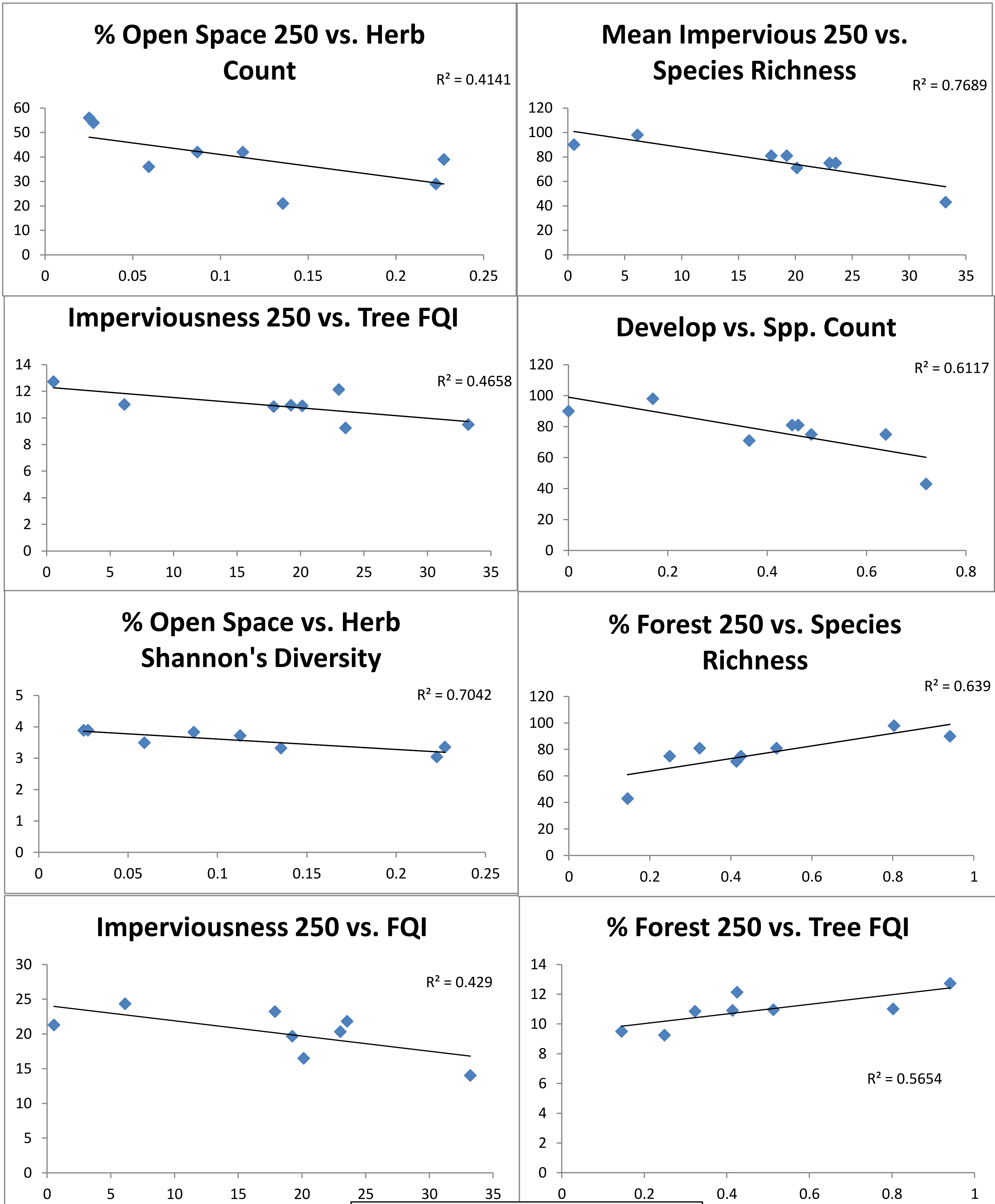


Figure 3: Ordination of study sites in overstory tree community composition space. NMS ordination with final stress of 9.55; final instability of < 0.01; Monte Carlo test P-value < 0.05. Vectors show the strength and direction of correlations between species cover and environmental predictor variables and axes. Only variables with significant R-squares (>0.40) are shown. Different plot symbols show groups derived from cluster analysis that differ in composition and environmental factors.



% Open Space 250 vs. Herb Count

Mean Impervious 250 vs. Species Richness

Imperviousness 250 vs. Tree FQI

Develop vs. Spp. Count

% Open Space vs. Herb Shannon's Diversity

% Forest 250 vs. Species Richness

Imperviousness 250 vs. FQI

% Forest 250 vs. Tree FQI

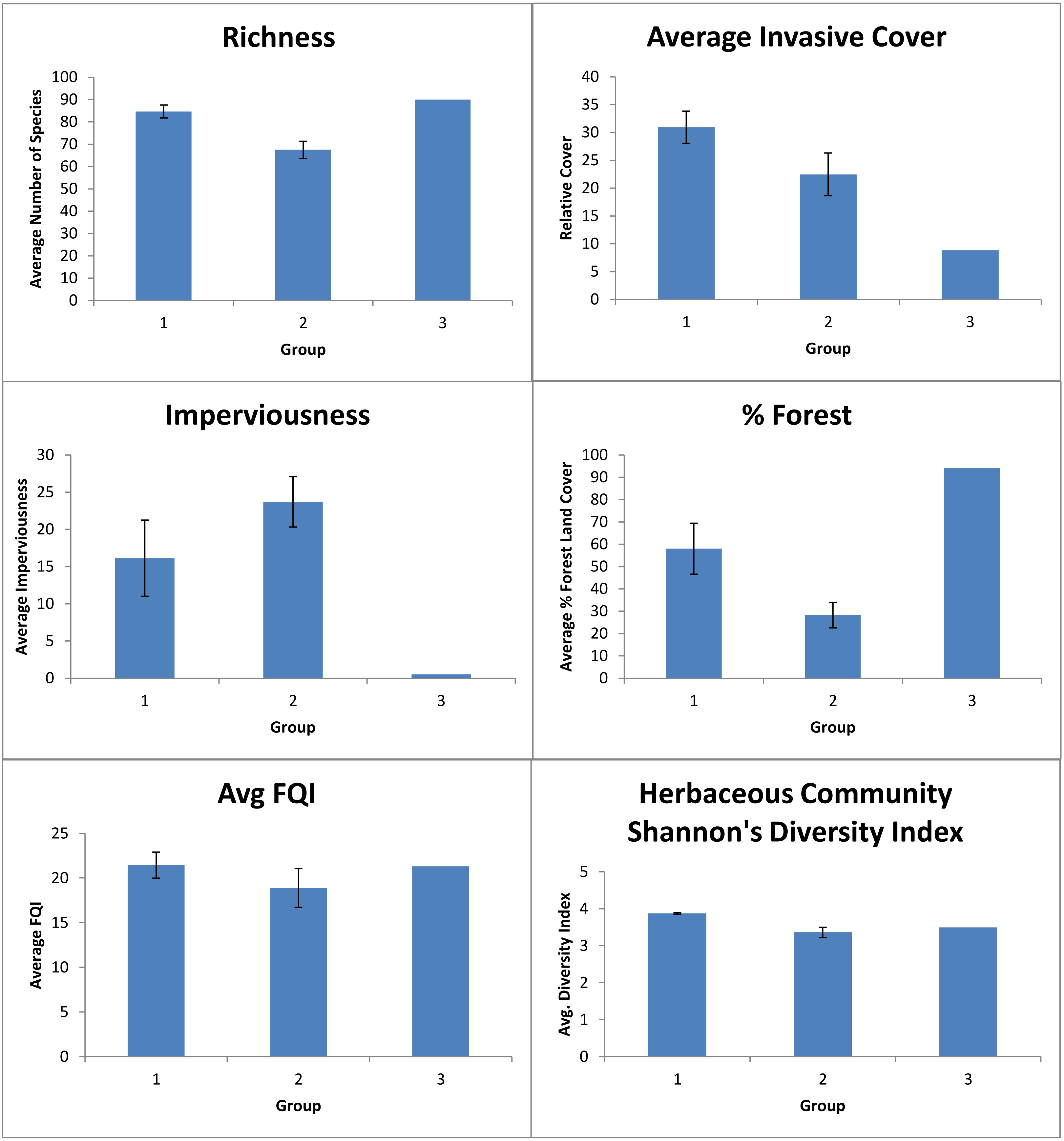
Panel 1: Summary of regression analyses of GIS predictor variables with significant correlations.

Group	Sites	Avg FQI	Indicator Species	IV	% Cover	p value	Coefficient of conservatism
1	1, 2, 13	21.44827	*Acer saccharum, Celastrus scandens, Duchesnea indica, Eupatorium maculatum, Ligustrum vulgare*	89.9, 82.4, 100, 92.4, 92.8	1.301, 4.74, 2.6369, 0.7815, 0.8698	0.0844, 0.0492, 0.0244, 0.077, 0.0492	4, 0, 0, 5, 0
2	4, 6, 7, 10	18.89252	*Acer negundo, Alliaria officinalis, Circaea quadrisulcata*	76.6, 64.1, 77.7	2.6744, 4.1136, 9.4782	0.1108, 0.0818, 0.0244	1, 0, 2
3	22	21.29267	*Apios americana, Laportea canadensis, Leersia spp., Sambucus canadensis, Sanicula marilandica*	NA	4.73, 32.88, 4.5, 6.3, 6.1	NA	3, 2, 3, 2, 6

Table 1a: Summary of indicator species analysis of herbaceous understory data.

Group	Sites	Avg FQI	Indicator Species	IV	Density	p value	Coefficient of Conservatism
1	1, 2, 22	21.77158	*Cornus racemosa, Ostrya virginica*	100, 100	0.008984, 0.019038	0.0276, 0.0276	2, 4
2	4, 7	15.26201	*Prunus serotina, Acer negundo, Cercis canadensis, Ailanthus altissima*	80, 70, 100, 62	0.014, 0.022348, 0.002682, 0.003242	0.0346, 0.1094, 0.0874, 0.1067	1, 1, 5
3	6, 10	22.52303	*Celtis occidentalis, Quercus alba*	67, 83	0.029009, 0.003839	0.0254, 0.1076	3, 5
4	13	20.32273	*Prunus serotina, Quercus velutina, Acer saccharum*	NA	0.0234, 0.01456, 0.02526	NA	1, 5, 4

Table 1b: Summary of indicator species analysis of tree canopy data


Richness

Average Invasive Cover

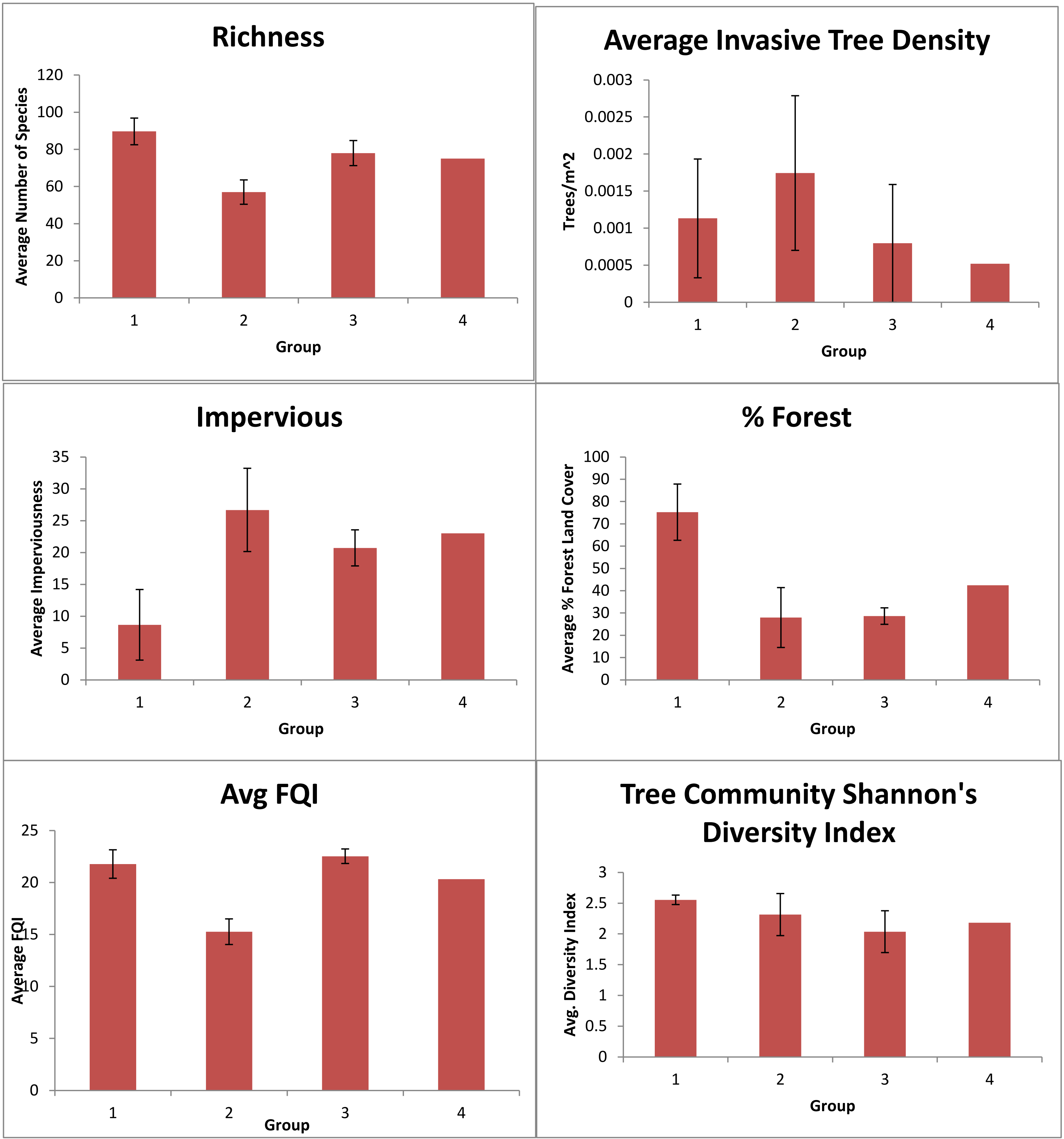
Imperviousness

% Forest

Avg FQI

Herbaceous Community Shannon's Diversity Index

Panel 2: Summary of average species richness, invasive cover, imperviousness, percent forest cover, FQI, and Shannon’s Diversity Index for herbaceous understory ordination groups.



Panel 3: Summary of average species richness, invasive cover, imperviousness, percent forest cover, FQI, and Shannon’s Diversity Index for tree canopy ordination groups.

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