# ASSESSING A RECONCILIATION ECOLOGY APPROACH TO SUBURBAN LANDSCAPING: BIODIVERSITY ON A COLLEGE CAMPUS

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ABSTRACT

As urban expansion continues to replace natural areas with non-native landscaping, native vegetation becomes increasingly scarce, and higher trophic levels that depend on native plant species decline, contributing to an overall erosion of biodiversity. The question addressed by this study is: Can reintroducing small patches of native habitat into suburban landscapes result in the subsequent recruitment of higher trophic levels of native biodiversity? We assessed plant, insect, bird, and small mammal biodiversity in four different habitats on the main campus of Calvin College in Grand Rapids, Michigan: open lawn, treed lawn, restored woodland plantings, and intact forest habitats. In four replicates of each area we evaluated plant, insect and small mammal diversity. We found that the restored woodland plantings had the highest diversity in each of the taxonomic groups. The lawn and treed lawn areas generally supported the lowest diversity, and the intact forest sites had intermediate diversity. We conclude that even small, relatively isolated islands of native habitat in a broader suburban landscape do have the capacity to increase abundance of higher trophic levels of native biodiversity.

KEYWORDS: biodiversity, restoration ecology, trophic interactions, habitat fragmentation, island biogeography

## INTRODUCTION

Biodiversity continues to decline globally as habitat loss and invasive species advance. While these two drivers of biodiversity loss have been well documented (Pimentel et al. 2004; Pimm et al. 1995; Vitousek et al. 1997), a more subtle aspect to the erosion of diversity in North America is the *way* we continue to develop our urban areas (Rosenzweig 2003). The traditional model of urban development essentially pushes the natural landscape out of the way, replacing it with a simplified topography and greatly reduced habitat diversity. Mostly non-native trees, shrubs, and turf grasses are introduced to accompany the newly built environment. The prevailing dualistic and misguided mindset that results is that 'nature' exists somewhere outside urban and suburban areas of human settlement and that the presence of human beings requires the sacrifice of native biodiversity.

Some non-native species planted in urban areas spread and become invasive

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in natural areas, which results in a diminished abundance of native plants and the associated loss of higher levels of native diversity dependent on native plant species. Non-native plants that are not aggressively invasive may still adversely affect biodiversity by being unpalatable to herbivores and unattractive to pollinators (Southwood et al. 1982; Tallamy 2004). Horticultural species that emerge through artificial selection (intentional or otherwise) can be more resilient to environmental stresses than native species, thereby pre-adapting them to a long-term presence in natural areas once they have been successfully dispersed (Torchin et al. 2003). Therefore, many non-native species originally introduced as ornamental plants have advanced into natural areas, either as harmful invasives or as innocuous waifs, contributing little or no benefit to higher trophic levels of biodiversity (Tallamy 2004). Furthermore, any time non-native plants are used instead of native species in landscaping, their presence (regardless of how invasive or persistent they may be) incurs a lost opportunity cost for the local biotic community.

Restoration ecology is one approach to address biodiversity loss, but this approach focuses on healing degraded ecosystems in an attempt to re-create more natural, and ecologically more functional habitats. While this is a worthwhile endeavor when appropriate, it is very expensive to do restoration well, especially on a large scale. In addition, most landscapes that have been altered from their original natural state are currently functioning to support human life and are not eligible even to be considered for restoration efforts (Hilderbrand et al. 2005; Hobbs et al. 2011).

Recent work in the areas of sustainability studies and urban ecology has helped establish a newly emerging paradigm for abating species loss—reconciliation ecology (Rosenzweig 2003; Pickett et al. 2008; Heffner and Warners 2011; Warners et al. 2014). Reconciliation ecology has been described as the practice of reintroducing native plants into urban and suburban environments to benefit native species (Rosenzweig 2003). The expectation is that these native plants will provide resources for insects and other species at higher trophic levels, thereby leading to an overall increase in native biodiversity specifically and strategically *within* the very places where high densities of human beings live, work, and recreate.

On the campus of Calvin College, in Grand Rapids, Michigan, this concept has been actualized in the form of four restored habitats on the main part of campus. Historically, this property was dominated by deciduous forests—oak and hickory on the better-drained, sandier soils, and beech and maple on the heavier clay soils. But the property was converted from forest to farmland around the turn of the twentieth century and later into areas of lawn interspersed with remnant woodlots and fencerows when the campus was created in 1957. In 2007, some areas of lawn were transformed into restored natural woodlands as mitigation for the loss of a mature oak–hickory woodlot on the campus. These restored areas were initiated with young trees 5–10 years old, some shrubs and herbaceous transplants, and locally collected seed mixes that were spread throughout each site. Species chosen for the restoration sites are all typically found in natural oak remnants in western Michigan and were likely present on the campus property prior to the original conversion to farmland. Local genotypes were exclusively used in these restoration plantings.

Although it is obvious that the restoration areas contain greater plant biodiversity than the lawns they replaced, a detailed evaluation of the vegetation that exists in these habitats had not been done since the original plantings were established. Therefore it was not known how much of the present plant diversity in these areas is due to desired native species and how much is contributed by unwanted non-native weeds. The context of multiple habitats existing within one campus provides a valuable opportunity to evaluate the potential of these restored areas to recruit higher trophic levels of biodiversity, a claim frequently made by restorationists yet seldom quantified.

Our approach was to sample the restored areas for plants, insects, birds, and small mammals and compare these data with identical sampling in three other campus habitats: lawn, treed lawn, and forest. If these areas of restored habitat yield greater biodiversity in higher trophic levels, it would indicate that even small islands of native landscapes can have a significant ecological benefit. However, principles of island biogeography would suggest a rapid diminishing return with smaller and smaller habitats, raising the question of whether benefits to higher trophic levels can be achieved with such small 'islands' of native habitat within a sea of suburban development. The hypothesis we tested has two parts: 1) that the restored and forested areas will each support significantly greater amounts of biodiversity than either the lawn or the treed lawn areas; and 2) that the restored and forested habitats will each support similar levels of native biodiversity.

## MATERIALS AND METHODS

***Location***

The Calvin College main campus is located in the southeastern portion of Grand Rapids, Michigan, and is bordered on the south and west by suburban residential neighborhoods. On the east side of campus is a college-owned 90-acre preserve that includes a mature woodlot surrounded by abandoned agricultural fields. Beyond this preserve to the east is a business corridor and interstate highway. To the north of the campus there is a mix of larger parceled residential lots and some scattered natural areas associated with the Reeds Lake drainage basin. The campus itself is dominated by lawn and treed lawn landscapes, as is typical of human-dominated suburban areas in the Midwest. A few small undeveloped forested remnants are interspersed within the campus landscape, as are the four restoration sites described above.

We collected data from four replicated 10 m × 10 m plots representing each of the four habitat types: lawn, treed lawn, restored woodland, and forested areas (16 plots in total). We defined "lawn areas" as open turf grass that is being actively maintained, and "treed lawns" as open turf grass maintained in the same way but containing at least one tree greater than 13 cm diameter at breast height and another tree of equal or greater size within 10 m of that tree. The forested areas used in this study were defined as current mid- to late-successional forest with no lawn and no maintenance other than the occasional removal of potentially dangerous snags and branches. The four restored areas are dispersed broadly across the campus and range in size from approximately 500 $m^2$ to 2000 $m^2$. They were all installed with a similar mix of native trees, shrubs, and herbaceous species. All 16 sites were selectively located to be at least 100 m away from each other (Figure 1).

Once these areas were identified, each site was mapped into as many 10 m ×10 m plots as they could contain. The plots were numbered, and one plot from each site was randomly selected for data collection. Plots were corner-marked with flags in the restored woodland and forested areas, while


FIGURE 1. Aerial view of Calvin College in Grand Rapids, Michigan, showing 16 study sites (4 replicates of each habitat type) as they are distributed across Calvin's campus.

in-ground markers were used for lawn and treed lawn areas. All plots were located at least 3 m from the edge of their respective habitats to minimize possible edge interactions.

***Plant Inventory***

To assess plant diversity, we randomly selected five 1 $m^2$ quadrats within each of the sixteen 10 m × 10 m plots. Within these five quadrats, we inventoried each species that was present and the relative percentage cover of each species (for a complete plant species list for all study sites please contact the authors). The sampling was carried out in all 16 sites during a three-week period in June 2011. From these data we were able to compare average number of species, relative abundance, and ratio of native to non-native species within and between habitat types (although these vegetative data will not be specifically reported in this paper).

During the fall of 2011 we also did a more comprehensive vegetation analysis in order to perform a Floristic Quality Assessment (FQA) of each of the 16 sites by recording all species encountered as we walked line transects at 2 meter intervals through each plot. This was first done in one direction and then in the perpendicular direction to ensure maximum coverage. From these lists we calculated a Floristic Quality Index (FQI) for each site using the coefficient of conservatism of each species, as assigned by the Michigan DNR (Hermann et al. 2001). A one-way ANOVA test was performed to compare mean FQI values among the different habitat types along with a Tukey-Kramer post hoc test to evaluate mean differences.

***Insect Biodiversity***

We collected insects by sweep netting both in summer and fall, covering all 16 sites at four different times—two in the summer (June 29–30 and July 6–13) and two in the fall (September 25 and October 8). On these occasions, we systematically swept through the tops of the herbaceous vegetative cover of each 10 m × 10 m plot. Insects trapped in the net were transferred to a jar of alcohol and later sorted in the laboratory. Relative abundance and length of each insect were recorded and a Shannon diversity index was generated for each habitat (Shannon and Weaver 1949; Magurran

1988). We tested the mean values by habitat with a one-way ANOVA and Tukey-Kramer post-hoc test.

### *Bird Survey*

To evaluate the frequency of bird visitation and use at our sites we conducted a bird survey at all 16 sites. The number and species of birds were observed during a 15-minute period at each site on four days in the spring of 2012 (April 21 and 29, May 3 and 17). The order in which these sites were visited was randomized to control for time of day as a potentially confounding factor. Birds that flew over the sites were not included, because direct use of the sites by the birds was the desired measurement. A Shannon diversity index was also generated for the bird data and a one-way ANOVA and Tukey-Kramer post hoc test was used to assess differences among the habitats.

### *Small Mammal Trapping*

To evaluate the distribution of small mammals among different habitat types, we conducted a small scale catch and release survey. We placed two Sherman traps in each habitat on July 20, 21, and 26, and August 10, 2011. The traps were baited with oatmeal, sunflower hearts, peanut butter, and a protein supplement and also contained a small wad of polyester fiberfill to guard against hypothermia. Two traps were set approximately 3 m away from each other in the middle of each 10 m × 10 m plot. The traps were set just before sunset in order to minimize the possibility of human interference. We checked all the traps before dawn the following morning and recorded species, sex, hind foot length, tail length, total body length and ear length for each individual captured. The tails were then marked with permanent marker for future identification. This procedure was done at all 16 test sites every time the survey was performed, resulting in a total of 128 trap-nights (16 test sites × 2 traps per site × 4 nights = 128 trap-nights).

## RESULTS

### *Vegetation*

Floristic Quality Assessment (FQA) is an evaluation of the floristic and natural significance of a given area based on native plant diversity (Herman et al. 2001). This significance is expressed in a calculated Floristic Quality Index (FQI), based on the mean coefficient of conservatism and the square root of the number of native species present. Since none of our lawn sites contained any native plants, we have no FQI to report for the lawn sites. The FQI of the restored woodland habitat was significantly greater than it was for either the forested habitat or the treed lawn (Figure 2). Although the forested habitat had a higher FQI than the treed lawn, this difference was not statistically significant ($0.05 < p < 0.10$). The percentage of native species in the forested and restored woodland sites was almost identical at slightly above 80%. The treed lawn habitat had a significantly lower native species component (22%), most of which was due to the presence of overstory trees, with relictual vegetation sometimes growing at the bases of the trees.

### *Insects*

The total average length of the insects from each site was calculated by taking an average length of all individuals of all taxa collected in a site. Four such values were generated for each habitat type, the total averages of which are re-

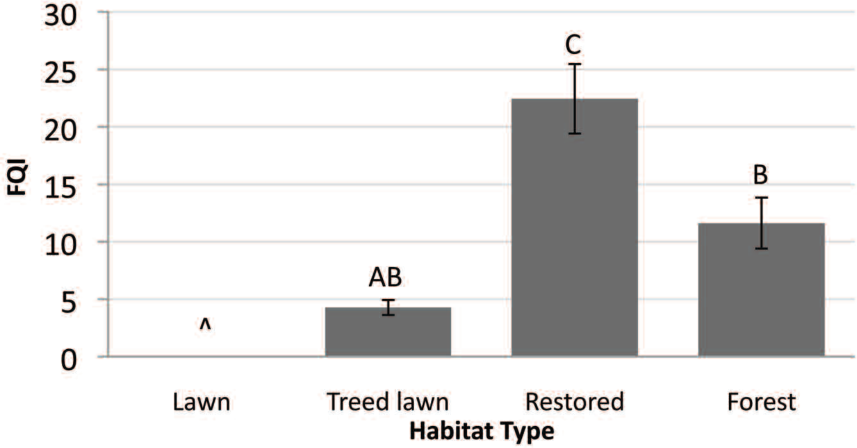


FIGURE 2. Average FQI for different habitat types. Bars not sharing the same letter are significantly different (One-way ANOVA, p<.05, n=4 for each habitat type). Error bars represent one standard error about the mean.

ported in Figure 3. The restored area was shown to have the highest average length ($p < 0.0001$), indicating that the largest insects are found there. The insects collected in the lawn and treed lawn sites had the lowest average length, and there was no significant difference between the averages calculated from

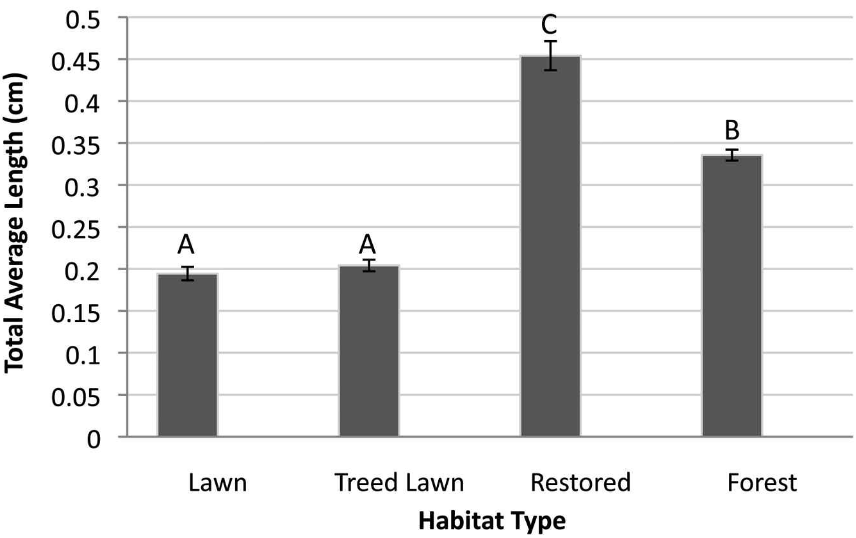


FIGURE 3. Total average length of insects in different habitat types. Bars that do not share the same letter are significantly different (One-way ANOVA, p<.05, n=4). Error bars represent one standard error about the mean.

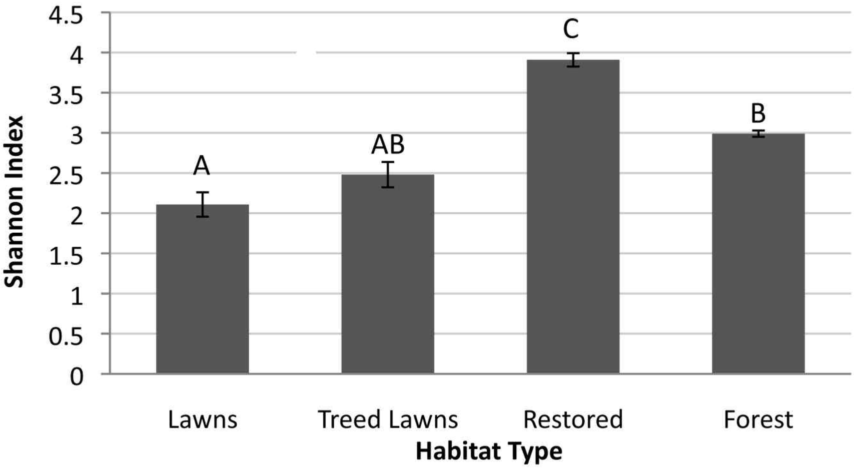


FIGURE 4. Average Shannon Index of insects for each habitat type. Bars that do not share the same letter are significantly different (One-way ANOVA, p<.05, n=4). Error bars represent one standard error about the mean.

these two habitat types. The insects collected in the forested sites had a higher average length than those from the lawn and treed lawn sites, but lower than that calculated for the insects in the restored woodland areas.

We calculated a Shannon index to quantitatively assess the richness and evenness in the diversity of insects in the different habitat types. Data collected from restored woodland habitats yielded a Shannon index of 3.67, which is significantly higher than that calculated for all the other sites ($p < 0.0001$) (Figure 4). The lawn and treed lawn sites have the lowest values, and they are not significantly different from each other. The forest habitat had an intermediate Shannon index, being significantly higher than the lawn areas, and significantly lower than the restored habitat, but not different statistically from the treed lawn.

### *Bird Survey*

We took an average of the Shannon indices of the four days of bird watching for each site, and then averaged those values within each habitat type. All of the average Shannon indices for each habitat type were less than 1, ranging from 0.82 for the restored areas to 0.04 for the lawns (Figure 5). The Shannon index for the restored woodland was significantly different from that for the lawn ($p = 0.017$), but was not statistically different from that for any other site.

### *Small Mammals*

We successfully trapped small mammals only in the restored woodland and forest sites. Although this part of our study was less extensive than the vegetation and insect sampling, in the 64 trap-nights for the lawn and treed lawn sites,

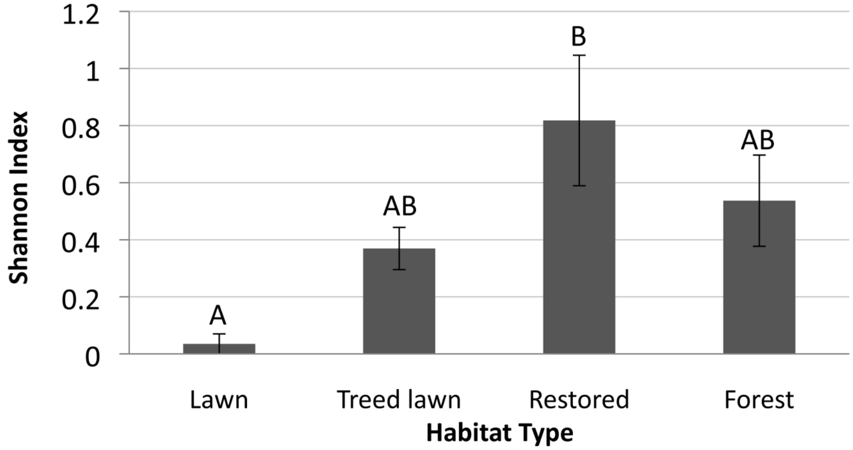


FIGURE 5. Total average Shannon index of birds in different habitat types. Bars that do not share the same letter are significantly different (One-way ANOVA, $p<.05$, $n=4$). Error bars represent one standard error about the mean.

TABLE 1. Inventory of small mammals trapped in Restored Woodland and Forest habitats (32 trap-nights per habitat). No small mammals were trapped in the lawn or the treed lawn habitat types.

| | Habitat Type | |
|---|---|---|
| | Species (common name) | Restored Forest |
| *Sorex cinereus* (Masked shrew) | 2 | 1 |
| *Peromyscus leucopus* (White-footed mouse) | 7 | 4 |
| *Peromyscus maniculatus* (Deer mouse) | 8 | 3 |
| *Microtus pennsylvanicus* (Meadow vole) | 3 | 0 |
| *Zapus hudsonius* (Meadow jumping mouse) | 0 | 5 |
| **Totals** | **20** | **13** |

we never caught a single animal. By contrast, in the restored woodland sites, we captured 20 small mammals of four different species in the 32 trap-nights, and in the forested sites we captured 13 small mammals of four different species (Table 1). The traps in the treed lawn sites sometimes showed signs of tampering (which could have been done by larger mammals, such as squirrels or raccoons), but, as noted, no small mammals were ever caught in these traps.

## DISCUSSION

This study evaluates the relative capacity of restored natural habitats (on the scale of approximately 1000 $m^2$) located within the context of suburban landscaping to support higher levels of native biodiversity. The data we collected support the hypothesis that both the restored woodland and forested areas will have greater biodiversity than either the lawn or the treed lawn areas. However, since most of our measures of biodiversity were highest in the restored woodland areas, we did not find support for our second hypothesis that restored and forested areas will support similar levels of native biodiversity.

The highest Floristic Quality Index (FQI) values were recorded from the restored woodland habitats, which had an average FQI value of 22.4 (Figure 2). Somewhat surprisingly, this value was nearly twice the mean FQI for the forested sites, which was 11.6. However, the restored woodland habitats had been planted only four years earlier and have been minimally maintained with occasional non-native removals and native species introductions. Because of the recent establishment of these areas, they support many young trees, a relatively high diversity of herbaceous perennials, and very little dense shade (Figure 6). The restored woodland sites are therefore similar to woodland edges, where both sun-loving and shade-tolerant plants can coexist (thereby elevating biodiversity) (Huston 1979; Leach and Givnish 1996). Furthermore, the forested sites are all relatively small (3,000–5,000 $m^2$), and, although they do provide dense canopy shade, they do not appear to be large enough or protected enough to support many of the more sensitive forest understory and ground-level species. Therefore, the biodiversity found in our campus forest sites is lower than that supported in similarly mature but larger tracts of forest in the vicinity. Yet based solely on the plants that are present in these sites, the Floristic Quality Assessment indicates that the restored areas represent the highest natural quality among these four habitat types.

By contrast, the lawn areas had an FQI of 0, because there were no native species found in any of these sites. The lawns are all actively managed and heavily dominated by Kentucky bluegrass (*Poa pratensis* L.) and some less abundant turf grass species, mostly because of consistent applications of broad-leaf herbicides. The treed lawn habitats had a higher average FQI (4.2) than the lawn habitats, both because of the presence of trees (most of which are native) and because some native herbaceous plants were found at the base of the trees, where they are able to avoid mowing and (apparently) herbicide application.

Insects are major pollinators and herbivores in terrestrial ecosystems, and they are the major food item for larger invertebrates, birds, and some small

FIGURE 6. Photograph of one of the restored woodland sites on Calvin's campus (van Reken Residence Hall).

mammals (Tallamy 2004). Other insects are vital components of the decomposing community. In short, insects are a major contributing element of a healthy ecosystem. Furthermore, several studies have shown that insects are associated with host-specific plants with which they co-evolve (Bernays and Graham 1988; Burghardt et al. 2008), underscoring the importance of native plant diversity for supporting native insect diversity.

This relationship is supported by the data we collected (Figures 3 and 4). The highest Shannon index for insects was calculated for the restored woodland areas (Figure 4). This pattern further supports our conclusion that the restored areas harbor the greatest ecological complexity. We found lawn areas, which had the lowest Shannon index, to be heavily dominated by only a few small-sized insect species, reflecting lower ecological complexity (Lawton et al. 1998). The treed lawn and wooded areas were not statistically different, which was surprising. Yet, in some of the wooded areas there was little to no ground cover, providing limited food sources for herbivorous insects. We suspect there are likely insects undetected by our sweep netting methods that reside in the soil, the bark of trees, and in the canopy that would increase the Shannon index value in the three habitat types that included trees.

Insect size diversity was also consistent with the plant data. Figure 3 shows the total average length of insects found in the four habitat types, with the largest value (0.45 cm) occurring in the restored areas. By contrast, average insect length in the lawn sites was 0.19 cm, significantly lower than that of the restored woodland sites. The presence of larger insects in restored woodland sites indicates the presence of higher trophic levels of insects there and likely indicates the presence of better food sources for insectivorous birds. Since 96% of birds rely on feeding insects to their young as a major protein source (Tallamy 2004), these restored woodland areas that support larger insects may well be providing important food resources for birds even beyond the more obvious benefit of seeds and fruit.

Considering the size of our sites and the relatively small amount of time spent collecting bird data, we still observed a large amount of bird activity. Data from restored woodland sites did produce a significantly higher Shannon index for birds than for the lawn areas. Although our small mammal sampling was even more limited, we find it noteworthy that small mammals were captured only in the restored woodland and forested areas (Table 1). The higher abundance and diversity of small mammals in these two habitats are likely due to the increased cover and food resources (plants, insects, and soil invertebrates). Together with our results of larger insects in restored woodland habitats, these bird and small mammal data provide further evidence that the restored woodland sites are capable of supporting higher trophic interactions. Although not assessed by this study, the higher abundance of small mammals in the restored woodland areas may provide subsequent benefit to predatory birds and terrestrial animals (anecdotally, we did notice that a garter snake has taken up residence in one of our restoration sites, and Cooper's Hawks are frequent visitors).

The consistent differences we report between restored woodland areas and lawn sites indicate that greater diversity at lower trophic levels (e.g., plants) supports greater biodiversity at higher trophic levels (e.g., insects, birds, and mammals) (Dyer et al. 2010). However, studies in island biogeography have shown that this basic ecological principle is limited by context and by scale (Darlington 1957; Gotelli 2008). The distance from a source site is one such variable that could be affecting the ecological interactions within our restored sites. Nevertheless, Watt et al. (2006) have reported that there can be rapid recovery of insect-plant interactions in restored areas up to 800 m away from a source location. Therefore, the successful recruitment of higher levels of biodiversity to our restoration plantings has likely benefitted from the presence of remnant natural areas in the vicinity, both on campus and in adjacent properties.

Island biogeography has also identified the size of a habitat as a determining factor for biodiversity. Even though our restored sites represent very small habitat fragments, they appear to be supporting significant populations at higher trophic levels. It would be helpful for future studies to address the benefit to biodiversity provided by urban restoration projects as the size of project and distance from remnant natural areas varies. It is highly likely that land use around such restoration projects is also a major influence worthy of assessment. Applying island biogeography principles to urban restoration and reconciliation ecology approaches will help provide a theoretical grounding for this newly emerging field (Pickett et al. 2008).

As institutions and businesses are increasingly looking for ways to decrease their carbon emissions, we propose that incorporating native habitats in their landscaping is a worthwhile consideration (Steensma et al. 2013). Such areas not only negate the need for fossil-fuel emitting activities (e.g., mowing, blowing, edging) and chemical applications, they also protect the soil, diminish stormwater runoff, and act as carbon sinks. In addition, as is evidenced by these data and documented by other studies (Burghardt et al. 2009), even small native habitats will support greater biodiversity. In an age of habitat decline and accelerated extinctions, any advances in biodiversity preservation are valuable and should be supported.

We encourage efforts to further understand the benefits to biodiversity from native habitat restorations, particularly in urban and suburban landscapes. If plantings like those on the campus of Calvin College were implemented across an urban landscape—school yards, church grounds, and municipal parks hold great potential for such initiatives—and an archipelago of native habitats were to emerge, the benefit to native wildlife could be significant (Bennet 1990). Reconciliation ecology efforts like these raise interesting and important research questions, particularly with regard to how higher trophic level interactions are influenced by the distance from the nearest source habitat, the size of such plantings, the vegetational diversity employed, and the broader land-use context within which the plantings occur. Principles of island biogeography are certainly implicated, yet when natural nature is reintroduced into such a highly human-dominated context, new trends and patterns likely await discovery.

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## LITERATURE CITED


Burghardt, K. T., D. W. Tallamy, and W G. Shriver. (2008). Impact of native plants on bird and butterfly biodiversity in suburban landscapes. Conservation Biology 23: 219–224.

Bennett, A. F. (1990). Habitat corridors and the conservation of small mammals in a fragmented forest environment. Landscape Ecology 4: 109–122.

Bernays, E. M. and M. Graham. (1988). On the evolution of host specificity in phytophagous arthropods. Ecology 69**:** 886–892.

Darlington, P.J. (1957). Zoogeography: The geographical distribution of animals. Wiley, New York, N.Y.

Dyer, L. A., T. R. Walla, H. F. Greeney, J. O. Stireman III, and R. F Hazen. (2010). Diversity of interactions: A metric for studies of biodiversity. Biotropica. 42: 281–289.

Gotelli, N. J. 2008. A primer of ecology. Sinauer Associates Inc, Sunderland, MA, USA.

Heffner, G and D. Warners (2011). Reconciliation ecology: A Christian pedagogy of place. Chapter 3 in Christine Fletcher, editor, Faith, Science and Stewardship: Christian Pedagogy on the Environment. Benedictine University Press, Lisle, Illinois.

Herman, K. D., L. A. Masters, M. R. Penskar, A. A. Reznicek, G. S. Wilhelm, W. W. Brodovich, and K. P. Gardiner. (2001). Floristic quality assessment with wetland categories and examples of computer applications for the state of Michigan, Revised second edition. Michigan Department of Natural Resources, Wildlife, Natural Heritage Program. Lansing, Michigan.

Hilderbrand, R. H., A. C. Watts, and A. M Randle. (2005). The myths of restoration ecology. Ecology and Society 10: 19.

Hobbs, R. J., L. M. Hallett, P. R. Ehrlich, and H. A. Mooney. (2011). Intervention ecology: Applying ecological science in the 21st century. Bioscience 61: 442–450.

Huston, M. (1979). A general hypothesis of species diversity. The American Naturalist 113: 81–101.

Lawton, J. H., D. E. Bignell, B. Bolton, G. F. Bloemers, P. Eggleton, P. M. Hammond, P. Hodda, et al.. 1998. Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. Nature 391: 72–76

Leach, M. K and T. J. Givnish. (1996). Ecological determinants of species loss in remnant prairies. Science 273: 1555–1558.

Magurran, A. E. 1988. Ecological diversity and its measurement. Princeton University Press, Princeton, New Jersey.

Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, P. M. Groffman, L. E. Band, C. G Boone, W. R. Burch, et al. (2008). Beyond urban legends: An emerging framework of urban ecology, as illustrated by the Baltimore ecosystem study. Bioscience 58: 139–150.

Pimentel, D., R. Zuniga, and D. Morrison. (2005). Update on the environmental and economic costs association with alien-invasive species in the United States. Ecological Economics 52: 273–288.

Pimm, S. L., G. J. Russell, J. L. Gittleman, and T. Brooks. (1995). The future of biodiversity. Science 269: 347–350.

Rosenzweig, M.L. 2003. Reconciliation ecology and the future of species diversity. Oryx 37: 194–205.

Shannon, C.E. and W. Weaver. 1949. The mathematical theory of communication. University of Illinois Press, Urbana.

Southwood, T. R. E., V. C. Moran, and C. E. J. Kennedy. (1982). The richness, abundance and biomass of the arthropod communities on trees. Journal of Animal Ecology 51**:**635–649.

Steensma, K. M. M., D. R. Clements, J. R. Wood, R. G. Van Dragt, and B. Lowe. (2013). Stewarding the gift of land: Christian campuses as land management models. Perspective on Science and Christian Faith 65: 104–115.

Tallamy, D. W. (2004). Do alien plants reduce insect biomass? Conservation Biology 18: 1689–1692.

Torchin, M. E., K. D. Lafferty, A. P. Dobson, V. J. McKenzie, and A. M. Kuris. (2003). Introduced species and their missing parasites. Nature 421: 628–630.

Vitousek, P. M. (1997). Human domination of the earth's ecosystems. Ecology 75: 1861–1876

Warners, D. P., M. Ryskamp, and R. Van Dragt. (2014). Reconciliation ecology: A new paradigm for advancing creation care. Perspectives on Science and Christian Faith 66: 221–235.

Watt, C. H. and R. K. Didham. (2006). Rapid recovery of an insect-plant interaction following habitat loss and experimental wetland restoration. Oecologia 148: 61–69.