

A STUDY OF BEETLE BIODIVERSITY
IN THE FORESTS, GARDENS, AND VACANT LOTS OF DETROIT

by

Julie Ann Cotton

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Thesis Committee:

Professor Ivette Perfecto, Chair
Assistant Professor MaryCarol R. Hunter
Assistant Professor Stacy Philpott*
Dr. M. Jahi Chappell*

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*Contact information for committee members not affiliated with University of Michigan:

Stacy M. Philpott, Assistant Professor
2801 W. Bancroft St., Mailstop 604
Department of Environmental Sciences
University of Toledo
Toledo, OH 43606

Dr. M. Jahi Chappell, Postdoctoral Associate
Science & Technology Studies
125 Rockefeller Hall
Cornell University
Ithaca, NY 14853

ABSTRACT

As the urban population of the world grows, ecologists are taking interest in the impacts of urbanization. Arthropods, and particularly carabid beetles (Coleoptera: Carabidae), are common focal organisms. However, beetle communities in shrinking urban areas are not well studied. This thesis compares the abundance, species richness and morphological composition of beetles captured in pitfall traps placed in remnant forests, community gardens and vacant lots of Detroit during May-August of 2007. The twelve study sites (four replicates of each habitat type) were also characterized using land cover and floristic measures. A total of 1039 beetles, including 183 morphospecies and 27 families were identified. Of the 199 carabid individuals of 30 species captured, over half were found in forest habitats. The large, carnivorous carabid beetles of forests are associated with open ground and canopy cover. The influence of environmental parameters indicated that the grassy ground cover of vacant lots is associated with a reduced capture rate of carabid beetles. In general, the urban habitats were dominated by flying species of carabid beetles, indicating that beetles that do not have the ability to migrate may have not survived the isolation of the site caused by urbanization. The forest of Belle Isle is notable because all the carabid species captured there are native to North America. The abundance and diversity of all the captured beetles was greatest in the urban gardens, which contain more abundant herbaceous and predatory beetles than gardens or forests. Gardens and vacant lots shared several species of beetles, while forest beetles were chiefly exclusive to forest habitats. Vacant lots were significantly lower in beetle abundance and diversity than either of the two habitats. A comparison of the beetle morphospecies, families and a subset of only ground-dwelling beetles (Coleoptera: Staphylinidae and Carabidae) found that the taxonomic grouping does affect the interpretation of the site similarities; the ground-dwelling beetle grouping appears to best represent the species diversity fostered by these habitats.

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PREFACE

THE GROWING CONCEPT OF URBAN IN DETROIT

The urban environment is inherently defined by human action, emerging from, and continually shaped by, anthropogenic factors (McIntyre, Knowles-Yanez, and Hope 2000). Though it may appear irrelevant in ecological studies set in “natural” areas, the history of an urban area, inclusive of social factors, is essential to a more complete understanding of the present state of the environment and may provide for a more constructive framing of the work for decision makers and land managers (Costanza et al. 2007). In order to adequately understand the current landscape of Detroit, developing a foundation of knowledge not only in ecological terms, but also of human social influences was essential.

The following material provides a review of some of the information I gathered about Detroit land uses – vacant lots, community gardens, and forests - that were used in the following study of Detroit beetle biodiversity. The concept and design of the study emerged from my concurrent education about the socio-economic challenges of Detroit that are documented in part in the beginning of this preface. Over the course of this project I have also come to understand that social and environmental inequities in urban areas are often sprouted from the same roots, causing systemic issues that deserve more than a reactive solution (Haughton 1999). I borrow from the research of other urban fields of study, including environmental justice, environmental psychology, urban planning and landscape architecture to consider grassroots movements that are working to address inequities and achieve sustainability in Detroit.

Challenges of Detroit

The industrial, racial, and land use histories of Detroit are tightly interwoven. This preface is not sufficiently detailed to capture the depth and complexity of the racial struggle faced by the black populations of Detroit (see Surgue 1996 for a detailed analysis), but in order to shed light on the “roots” of the Metro Detroit area crisis, I offer this rudimentary summary.

Many black migrants came to Detroit to escape the limitations of southern society beginning in the early 1900s. Upon arrival, they systematically faced exclusion from the housing market, stable and fair employment, and participation in governance (Sugrue 1998). After decades of a multitude of injustices, the largest and most deadly racial riot in U.S. history took place in the summer of 1967. The movement of industry to the urban outskirts and the blockbusting activities of exploitive real estate practices, carabid were initiated before the riot, burgeoned by the perceived threat of racial violence following the riot, encouraged the relatively more-affluent white families to flee to the suburbs (Sugrue 1998). Limited ability to accumulate personal wealth in the form of wages or property resulted in the black population being largely unable to follow jobs that were leaving the city.

The population of Detroit has declined continuously since it's peak of 1,850,000 in 1950 (Gibson 1998), and currently numbers 834,000 (U.S. Census Bureau 2006). The Detroit metropolitan area remains one of the most racially segregated regions in the country, with the city comprised of approximately 80% black citizens and 12% other minorities, while the surrounding suburban areas are predominantly white (Iceland, Weinberg, and Steinmetz 2002). Detroit's rapid expansion through the Gilded Age of shipping and the Industrial Revolution produced a sprawling city that encompasses approximately 139 square miles (Gibson 1998).

Unemployment

Unemployment plagues Detroit and has brought about an incomparable socio-economic situation. Over one third of the residents currently live below the poverty line, a greater number than any other large city in the U.S. (U.S. Department of Labor 2008). The Detroit metropolitan area has maintained one of, if not the, highest unemployment rates in the nation since 2004, and has been topping the chart in job losses beginning in 2001 (U. S. Department of Labor, Bureau of Labor Statistics 1999-2009).

Vacancy and Blight

The significant loss of Detroit business and residents has resulted in the vacancy of approximately 66,000 parcels, or 40 square miles, throughout the city (SEMCOG 2003). In the midst of the current housing crisis, Detroit also had the highest rate of metropolitan area

home foreclosures in the nation during 2007, with almost five percent of its households entering a stage of foreclosure during the year (RealtyTrac Staff 2008).

However, delinquent taxes may be as much a reason for foreclosures as risky lending practices, as the remaining residents and homeowners must support an infrastructure that once served twice the population through taxes and high utility charges. Abandoned parcels that are now city-owned resulted in a loss of tax revenue that amounted to \$95 million in 2002 (Detroit 2002). Of course, vacant properties exacerbate the municipal fiscal impacts by decreasing the property values of surrounding holdings, thereby further stifling tax revenues and requiring increased public services such as fire and police attention due to greater public safety problems associated with vacant structures (Kraut 1999).

The presence of vacancies in the City of Detroit has resulted in an astounding re-vegetation phenomenon. In contrast to other urban vegetation studies that correlate increasing vegetation with greater socioeconomic means (e.g. Harlan et al. 2007), Ryznar found that social stress and the subsequent neighborhood decline, marked by a low median income and other quality-of-life factors is associated with and can be distinguished by increased vegetational growth within the city (1998). Declining neighborhoods are punctuated by large vacant open spaces where abandoned homes or buildings have been demolished (Ryznar 1998). Ruderal grasses, remnant ornamental plants and trees, and a few hardy herbaceous plants dominate most open lots in Detroit, and are maintained at the expense of the city by occasional mowing.

Struggling School System

An additional \$67 million in unpaid taxes was owed to city schools as of 2002 (Reece 2004). The Detroit Public Schools system has closed approximately 60 schools, and equally as many schools are operating at only a portion of their capacity as students move from the city or to alternative educational systems (Dvoboda 2007). Many community members and government officials feel that the communities are “held together” by neighborhood schools, and that closures will result in a loss of this “bond” and further relocation out of the city (Dvoboda 2007). This retraction, reportedly one of the most extensive in the nation, transformed the district into one of the largest landlords in the city, as most of these dilapidated school buildings are difficult to sell (Dvoboda 2007).

Urban Tree Loss

Once known as “tree town” (see Figure 1) for the towering trees that lined the residential streets, Detroit lost half its tree canopy to Dutch elm disease, development, and poor maintenance between 1950 and 1990 (Detroit Free Press 2006, in American Forests 2006). The origin of the Emerald ash borer infestation was near Detroit, and more than 30 million ash trees in southeastern Michigan have died



Figure 1. A picture of the downtown area of Detroit in 1942 [as seen from the famous Fisher building facing northwest] (National Photo Archives)

or been preventatively removed, further reducing the canopy (Michigan State University, State of Michigan and USDA *Emerald Ash Borer Information* 2004-2008). High resolution satellite data obtained for Detroit in 2005, before the effects of Emerald Ash borer were fully

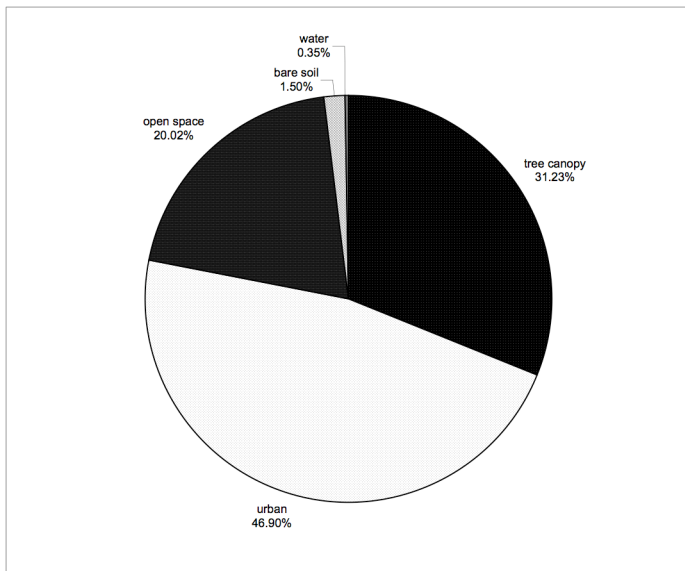


Figure 2: percent land cover in the city of Detroit encompassing approximately 370 square kilometers, as determined from satellite data (adapted from American Forests 2006).

visible, shows that Detroit is primarily covered with impermeable urban surfaces, but that tree-covered areas and open space without trees account for the majority of the remaining land cover (American Forests 2006) (Figure2). The Detroit and Rouge Rivers have been identified as “Areas of Concern” due to the sewage overflows that impact these rivers and subsequently the Great Lakes, originating from the loss of green infrastructure during the urbanization of in the sprawling

metropolitan areas surrounding Detroit (EPA 1998, in American Forests 2006).

Unfortunately, the city is also faced with the fact that it no longer has the financial resources to maintain several of the smaller urban parks throughout the city that are in poor condition. The city is considering selling parks in areas with high population loss in order to better maintain parks that serve greater populations or are larger in size. Of course, this sale and subsequent alternate use may cause further loss of available greenspace in underserved areas, and homebuyer interest in the area could be further reduced (Saulny 2007).

Food Insecurity

Studies evaluating food availability in Detroit have shown significant lack of access to fresh produce and healthy foods in grocery retailers, resulting in a “Food Desert” (Pothukuchi and Thomas 2004; Mari Gallagher Research and Consulting Group 2007). There are few if any large supermarkets within the city boundaries of Detroit, as the remaining large-scale supermarket retailer closed its remaining outlets in July 2007 (Pothukuchi and Thomas 2004; Wilczynski 2007). Over half the current population lives in areas that are at least a mile from the nearest grocer of any size with a healthful selection of food (Mari Gallagher Research and Consulting Group 2007). One study quantified that only eighteen percent of the stores selling food in three low-income zip code areas of Detroit were found to have a minimal “healthy food basket” of items necessary to assemble a balanced meal (Pothukuchi 2001).

Public health researchers in the city have found that access to fresh foods correlates most directly with socio-economic status. Research considering the race and socioeconomic status of the Detroit residents found that African Americans in middle-income communities have similar access to supermarkets as their white middle-class cognates (Zenk, Schulz, Israel et al. 2005). A study of African American women’s choice of food markets and perception of the quality of the available fruits and vegetables found that more educated and higher income women were able to access stores that offered better stocks of fruits and vegetables, and therefore consumed more of these items (Zenk, Schulz, Hollis-Neely et al. 2005).

The Mari Gallagher Research and Consulting Group (2007) conducted an intensive survey of food outlets on a block-by-block basis, discovering startling correlates with

physical health outcomes. About 550,000 Detroit residents, over half the current population, live in areas that are at least a mile from the nearest grocer with a healthful selection of food, while the nearest convenience store or other “fringe” market is, on average, 0.2 miles from their residence. This imbalance in food availability is related to greater incidence of illness and premature death from diet-related conditions such as diabetes, cardiovascular disease, cancer, hypertension, and obesity. Groups such as single mothers, the elderly, and diabetic patients have greater difficulties in meeting the nutritional recommendations for health maintenance due to a lack of access. Young, low-income African American mothers have significantly increased odds of depression due to food insufficiency and deteriorated housing (Siefert et al. 2007).

Fifty-six percent of the Electronic Benefits Transfer (EBT, formerly the Food Stamp program) retailers were found to be party stores, dollar stores or other fringe venues. Only eight percent of stores that accept EBT were recognized by the study as grocers or supermarkets that offer healthful food. These fringe venues are the largest contributors to the negative effects of the food imbalance, suggesting that those in economic hardship are the most likely to be affected by the lack of access to healthy foods. These factors contribute to premature and excess death among residents living in areas with significant “food imbalances”, meaning that access to less nutritious food is much easier than their access to fresh, nutritious foods. (Mari Gallagher Research and Consulting Group 2007)

The most greatly imbalanced neighborhoods are inhabited by African Americans, which on average are 1.1 miles further from the nearest supermarket than were white neighborhoods (Zenk, Schulz, Israel et al. 2005). In low socio-economic African American communities, liquor stores are four times as prevalent as they are in middle-income, racially heterogeneous neighborhoods, with a corresponding lack of true grocery stores (Zenk et al. 2006). The higher use of food assistance programs like EBT in poor neighborhoods means that purchasing is concentrated to the days when the benefits are distributed; thus, the EBT form of payment often leads merchants in these neighborhoods to limit fresh produce availability only to coincide benefits distribution.

Lack of Transportation Options

Transporting people to food, jobs and other amenities has many limitations in Detroit. In the city that is known as the center for the automotive industry, owning and driving a vehicle is often unfeasible. Low-income households are six to seven times more likely than other U.S. households to not own cars (Murakami and Young 1997; in Vallianatos, Shaffer, and Gottlieb 2002). Ironically, Motown has the highest insurance rates in the nation, making car ownership more expensive than in any other city (CNNMoney.com 2006). The lack of money for road maintenance and repair often leaves a maze of roadways that are largely unavailable to drivers. Currently, the City of Detroit has little in the way of public transit that serves the people who want to get from the inner city area to the suburbs. Bus routes, the sole type of public transit in Detroit at the moment, currently only run within the city, and suburban transit lines are also limited within separate boundaries (Transportation Riders United 2006-2008).

Industrial Legacy

Much of Detroit also bears the scars of its industrial legacy— the pollution from decades of machining and chemical work. Thousands of properties, including those under city ownership, are identified as “brownfields”. Brownfields are defined as “a property, the expansion, redevelopment, or reuse of which may be complicated by the presence or *potential presence* of a hazardous substance, pollutant, or contaminant” [emphasis added] (U.S. Environmental Protection Agency 2008). A definitive number or area of brownfield sites is not available, as the designation of “brownfield” is generally determined only when properties are investigated prior to sale or redevelopment. However, there is some hint to the magnitude in the records of the Michigan Department of Environmental Quality: at least 3000 business or former businesses in Detroit are potential brownfield sites because they have utilized or accepted controlled wastes (MDEQ 2008 a.); 767 leaking undergrounds fuel storage tank sites have not yet been remediated (MDEQ 2008 b.); and 203 “state-nominated and state-funded cleanup sites as well as sites that have been redeveloped using the Baseline Environmental Assessment process” currently posted in the Wayne County brownfield database (MDEQ 2008 c.). Considering that potential contamination - such as illegal dumping or air or water drift from industrial processes - can be reason for brownfield

designation, it is arguable that the majority of the city could be eligible for the brownfield programs.

Complicating the matter of housing is the fact that over 90% of the city housing stock was constructed before the lead paint law was passed in 1978; lead based paint is primary cause of lead in the human environment (U.S. Environmental Protection Agency 2007). The practice of demolishing standing dilapidated buildings by packing the building materials such as pipes and painted facades into the former basements has contributed to lead in the soil throughout Detroit, making residential areas possible sites of contamination even when the house is no longer visible. Lead poisoning affects approximately 10% of the children under the age of six in Detroit, a rate that is more than twice that of the rest of Michigan (Thompson 2002).

Considering the racial disparities and the multitude of environmental challenges that face Detroit, it is logical that the Environmental Justice (EJ) movement partially emerged through research conducted with the city. University of Michigan Faculty Investigators Paul Mohai and Bunyan Bryant lead the first comprehensive examination of environmental inequality in Detroit area in 1990 (Bullard et al. 2007). There are 12 hazardous waste facilities within the metropolitan Detroit area; only Los Angeles, with 17 locations, has more facilities within a metropolitan area (Bullard et al. 2007). *Toxic Wastes and Race at Twenty: 1987-2007*, the recent update of the landmark environmental justice report, found that Michigan has a greater disparity in the number of people of color living near hazardous waste sites than any other state in the nation. In particular, 69.3% of the people in the host neighborhoods of the Detroit metropolitan area are “of color”, with the greatest burden on the African American community, accounting for 60.8% of the total population (Bullard et al. 2007).

Addressing Challenges

The magnitude of the problems in Detroit can be overwhelming from the perspective of an observer, but these issues are real obstacles to health and well being for the residents of Detroit. Yet community development efforts, aided by governmental contributions and other social programs, are beginning to achieve results.

Advocacy groups such as the Sierra Club's Detroit Environmental Justice Program and Detroiters Working for Environmental Justice have brought attention to environmental issues, and forced municipal action to reduce environmental hazards (Bullard et al. 2007). For instance, the groups pressure helped to end the use of the Greater Detroit Resource Recovery Facility waste incinerator, in part due to studies that reported higher asthma rates near the incinerator. The Detroit Area Studies [DAS], initiated in 1951, are a series of social surveys that explore various pertinent issues through respondent surveys. These studies, conducted on an almost annual basis, represent a large data set that complements the conventional EJ considerations of pollution risks by adding the perception of the residents in response to their environment.

The EJ movement certainly does not lack for substantive issues regarding pollution and health risk in Detroit. However, the field of EJ has begun to reshape its efforts to address not only environmental hazards, but also the lack of amenities such as access to fresh and nutritious food, transportation, and safe areas where people can interact with nature in urban settings (Bullard 2007). Increasingly, international organizations such as the Food and Agriculture Organization of the United Nations and researchers are identifying access to nutritious food and nature as more than privileges, but essential aspects of life that should be considered human rights (FAO Voluntary Guidelines to support the progressive realization of the right to adequate food in the context of national food security. 2005; Urbana-Champaign 2009).

Planning and Design

The cultural constructs and economic trends of a community shape – and are shaped by - the physical landscape (Nassauer 1995). In urban areas, the practitioners that most often shape the environment include urban planners and landscape architects. With limited funds available, though, the city planning office is working with a number of organizations and academic institutions to address the realities of negative growth.

The nonprofit public policy think-tank The Brookings Institution is recognized as a leader in downtown revitalization efforts and has considered the redevelopment of downtown Detroit (Brookings Institution Metropolitan Policy Program et al. 2006). However, the basis of their twelve-step revitalization plans (Leinberger 2005) are to building corporate

livelihood, not livable space for current residents. Unfortunately, this means that their suggested first physical developments, entertainment venues such as the new Tiger Stadium and casinos, are for courting those with disposable income, while developing a local retail market is postponed until the more wealthy former suburban residents have been lured to populate the area.

In contrast, the landscape architecture ideas incorporated into “new urbanism” design provide for self-sufficient communities that offer basic amenities within walking distance. Such developments are intended to foster community interaction and incorporate shared greenway or garden areas and other features that allow community members to have access to nature. Though no areas of Detroit have been designated for sustainable redevelopment by municipal governance, art communities and local shops in the Cass Corridor (Midtown) area have sparked thriving new businesses and the construction of condominiums in former commercial sites. Another artist couple in the northern Detroit drew a rash of media attention and home buying near their Powerhouse Project, a solar-powered artist-in residence venue (Wasacz March 17, 2009). A pair of architects, Greg and Vibeke Vendena work on phytoremediation, material reuse and sustainable house renovations (Vendena and Vendena 2009). These are only a few examples of the grassroots efforts among existing and new residents of Detroit that preserve elements of the cultural legacy of the city while encouraging community reinvestment.

Efforts to expedite the redevelopment of vacant lots have considered ways that properties could be bought for minimal fees in order to restore owner maintenance and tax revenues. Perhaps the most recognized of the community organizations working on this issue is the Metropolitan Organizing Strategy Enabling Strength (MOSES), which is forwarding the concept of a municipal land bank, integrated with the City of Detroit Planning Department that would sell vacant land parcels for a minimal fee (Reece 2004). Land banking could allow for individuals without access to large amounts of capital, non-profits, and other small business entrepreneurs to contribute to community redevelopment.

Replanting Urban Trees

Green infrastructure such as trees and vegetative land cover influence stormwater management, water quality and air quality factors. Even with man-made stormwater

structures, several water contaminants are known to increase as the purifying function of the green infrastructure degrades. Based on the stand data of 2005, American Forests quantified the services provided by the trees of Detroit annually: stormwater management of 191 million cubic feet of water, valued at \$382 million; removal of 2.1 million lbs of air pollution, valued at \$5.1 million annually; and sequestration of millions of tons of carbon (2006).

The presence of trees and common vegetated areas in low-income residential areas leads community-building through informal social interaction, and is correlated with a reduction in crime (Kuo et al. 1998; Kuo and Sullivan 2001). Well-managed trees and grass provide a sense of safety (Kuo, Bacaicoa, and Sullivan 1998; Kuo 2003), while aesthetic benefits of urban trees and vegetation can assist in creating a setting that is more likely to encourage emotional attachment of residents to their neighborhood (Perkins 2004). Besides being an attractive and beneficial element among private homes and businesses, trees provide environmental services such as reducing energy use in structures (Nowak and Crane 1998; Wolf 1998 a.). For example, a 25-foot tree near a typical home will reduce the annual heating and/or cooling costs by 8-12 %, and a mature canopy of trees can reduce local air temperatures by about 5-10 °F (Wolf 1998 b.). Trees reduce smog levels by 6% simply through a reduction in air temperature (Wolf 1998 b.).

In a city that struggles with crime and meeting educational needs, giving attention to urban forests may be a way to foster beneficial outcomes. Integrating the natural and socio-cultural environment into school curriculum has been shown to provide important learning benefits. A study of diverse schools that integrated their curriculum with the surrounding environment found a reduced need for discipline, increased student engagement and enthusiasm, greater student pride in accomplishments, and significant gains in test scores related to all academic subjects (Lieberman and Hoody 1998). By providing a vegetated area for exploration, students can apply otherwise abstract concepts in a tangible setting, increasing their ability to retain new knowledge. Children with attention deficit disorder experience reduced symptoms and increased concentration after exposure to green play areas (Taylor, Kuo, and Sullivan 2001).

The Greening of Detroit nonprofit, founded in 1989, has been helping volunteers and residents replant trees in parks and neighborhoods that were impacted by tree loss. Recently,

the city agreed to let Greening of Detroit redevelop the 125-acre Walter Meyers Nursery site in northwest Detroit to supply trees for its program and to provide a venue for outreach programs (Pennefather 2009).

Community Food Security & Urban Agriculture

The emerging movement in community food security is defined by the Community Food Security Coalition as “a condition in which all community residents obtain a safe, culturally acceptable, nutritionally adequate diet through a sustainable food system that maximizes community self-reliance and social justice” (Hamm and Bellows). Professionals focusing on the Community Food Security movement have also courted the more politically powerful EJ movement by recognizing shared challenges and objectives in serving the malnourished urban community (e.g. Gottlieb and Fisher 1996). The food desert research outlined in the challenges section confirms that food insecurity exists in Detroit, and that providing greater access to fresh nutritious produce would be of value in the city. There are two commonly considered ways to address the lack of access to food. One is finding a way to transport people to existing markets; the other is to produce the food in the area where it is needed.

Despite the difficulties, most low-income households attempt to use cars for food shopping, even though more than half cannot rely on a car that they own (U.S. Department of Agriculture 1999). While traveling produce distribution vans have been tried in other urban areas, such efforts require maintaining a strong volunteer base and other external inputs, such as vehicle fuel, that may not be sustainable, especially in times of high fuel prices with a dispersed population of families in need. Creative public-private partnerships that provide transportation to supermarkets (Vallianatos, Shaffer, and Gottlieb 2002) might be a stopgap measure, but considering the massive area of the city, this scenario would be unlikely to be able to serve all the populations in need. Both the environmental and economic sustainability of such a transportation modification approach is questionable; as such, it is unlikely to provide long-term community food security according to the definition given above. Continued lack of community-based business, reasonable cost, and availability at the local level is unlikely to improve if residents are dependent on the stores to provide transportation in order to meet their basic needs. Residents that operate on short-term budgets due to day-

labor jobs or fiscal necessity, such as the elderly, purchase food when money is available and may not be able to take advantage of such a program.

Another option for addressing the Detroit food desert issue is to produce fresh produce locally through urban agriculture. This option requires agricultural resources and season-extension expertise, but it is a more self-sustaining concept and has additional benefits that could bolster well being. Though a number of lots may be unsuitable for agriculture due to pollution, most of the areas of the city were first settled as agricultural lands (Gavrilovich & McGraw 2001, p.30) and have potential to be returned to this use. Urban agriculture has a long history in Detroit, with peaks in public and private support during times of great economic stress – times not unlike today. The city of Detroit claims to have created the first municipally-supported community gardening effort in the U.S. when it allotted 455 acres of land and seed potatoes for planting to 945 families during the economic depression of the mid-1890s (Hanna and Oh 2000). The city's program of temporary leasing of abandoned land was replicated in more than 20 cities in the US, but with the increase in real estate development these gardens were short-lived (Hynes 1996). The next revival of urban gardening came with the “liberty gardens” of World War I and then the postwar “victory gardens” (Hynes 1996). In 1976, Detroit was the recipient of one of five federal grants administered by the USDA Cooperative Extension Service to start an Urban Garden Program that provided teaching and demonstration garden projects; the program ceased in the early 1990s (Gopakumar 2005). The near demise of the industrial paradigm in the mid 1990's sparked the current renaissance of community gardening in Detroit, supported by organizations such as the Boggs Center to Nurture Community Leadership (BCNCL) and the Detroit Agricultural Network (DAN), a branch program of Greening of Detroit (Gopakumar 2005).

A Community Food Assessment of Detroit conducted by Pothukuchi of Wayne State University (2001) identified “salient” issues among community and food activists to include 1) food and community economic development, 2) neighborhood revitalization, 3) community health, hunger and food insecurity, and 4) regional agriculture. Urban agriculture holds promise in contending with these issues. The DAN – affiliated Garden Resource Program Collaborative has helped over 500 backyard, community and school gardens through educational and material support (*garden resource program collaborative: garden resource*

program 2009). In addition to increasing their own food access, these small-scale farmers are marketing their produce at rotating farmer's markets throughout the city, and have their own stand and "Grown in Detroit" marketing program available at the historic Eastern Market in Detroit. Urban agriculture can also provide alternative sources of income and job training opportunities, community participation, waste management (compost) and grey water use opportunities within a city system (Pinderhuges 2004). Research in New York City determined that community gardens have a positive impact on the value of residential properties within 1000 feet of the garden (Been and Voicu 2006). This impact increases over time, with the greatest benefit in the "most disadvantaged" neighborhoods, with the quality of the garden positively correlated with the level of the impact. Founding a garden increases rates of homeownership, and "thus may be serving as catalysts for economic redevelopment of the community" (Been and Voicu 2006). The potential health benefits of urban agriculture are also encouraging. In addition to food security, participants are rewarded with improved nutrition, exercise, mental health, social and physical urban environments (Bellows, Brown, and Smit 2003).

The existence of these gardens speaks to the desire of Detroit residents to address their needs internally, but the demise of the several former programs also speaks to the challenges of sustaining urban agriculture. When the value of land increases, gardens can be lost to developers. However, the multi-faceted support provided by Detroit Agriculture Network recognized this issue, and is working towards creating a land trust for the community gardens that wish to participate. The city is also granting the purchase of 2.5 acres adjacent to the Eastern Market for a small farm that will provide produce to the market.

Concluding Remarks

While the demographic groups affected may differ, the post-industrialization decline and subsequent social challenges experienced by the City of Detroit are replicated in declining "rust belt" areas throughout the United States. Although the intensity of deterioration is currently unique to Detroit (Sugrue 1996, p.3, 13-14), Detroit may serve as a bellwether for other declining urban areas.

The multi-dimensional and multi-layer nature of socio-economic, racial, and environmental inequalities and inequities often leaves residents of Detroit, and especially the

youth, seemingly bound by the challenges of their environment. The lack of a strong education system proliferates unemployment, and poverty leads to enduring sub-standard housing and food systems that further reduce the well being of individuals. The starved and bleeding tax coffers of the city result in fewer amenities, and the ability of the population to maintain their mental and physical health is limited.

The cultural constructs and economic trends of a community shape – and are shaped by - the physical landscape (Nassauer 1995). Solutions in the form of reforestation, urban agriculture and restoration appear to be ready to address the issue of this shrinking industrial city in simple ways that the Detroit municipal governance has not yet seized upon. The sustainability and survival of urban areas such as Detroit is dependent on the self-reliance of the local economy. Planting trees and gardens, and fostering creative entrepreneurial activities are relatively “low-hanging fruit” of community development – they are low cost and don’t require a significant capital buy-in by the community or government. New cultural constructs that liberate the city’s economy from the industrial paradigm have most certainly taken root, and they are shaping a sustainable future for Detroit.

I hope that assembling this information in my thesis may provide insights for other urban ecological studies in areas experiencing similar elements of decline. By initiating studies of the biodiversity in Detroit, I wanted to not only add to the body of knowledge in urban ecology, but also to contribute to the informed conservation, restoration, and maintenance of forests and gardens in Detroit.

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CHAPTER I

INTRODUCTION

The increasing recognition of the influence of humans in the environment has brought about a “paradigm change” in the last few decades that moved conservation biology from a singular focus on unmanaged “natural” habitat to one incorporating the role of human-influenced disturbance, with agricultural and urban systems becoming new foci of ecological research (McDonnell 1997; Vandermeer and Perfecto 1997). Although urban areas cover only three to four percent of the world’s area (UNDP, UNEP, World Bank, World Resources Institute 2002), cities are now the home to more than fifty percent of the human population, and are expected to continue to grow to 70 percent of the world population by 2050 (United Nations 2008). The continued growth of small cities in developing nations, and to a lesser extent larger metropolitan areas (United Nations 2008), in the coming decades represents a challenge for conservation and an opportunity for implementing new research and land use concepts (Musacchio et al. 2005; Liu et al. 2007). Species richness of all taxa tend to be reduced in highly urbanized areas, while suburban areas show mixed responses with trends of increasing flora and birds but decreasing invertebrates and non-avian vertebrates (McKinney 2008). Elements intrinsic to human disturbance such as the introduction of nonnative species, creation of a more heterogeneous environment, level of disturbance and variability in the patch scale may all influence species richness, which makes for complex *in vivo* studies that often have conflicting results (McKinney 2008).

Though it is often overlooked, there is also a need to understand the dynamic role of humans in forming the landscape (Odum 1969; Nassauer 1995; Grimm et al. 2000; Turner, Nakamura, and Dinetti 585; Pickett et al. 2001), as reinstating this relationship may allow for the simultaneous improvement of human well-being in both economic and psychological terms (e.g. Kaplan and Kaplan 1995/1989; McKinney 2002; Kuo, Bacaicoa, and Sullivan 1998; Kuo and Sullivan 2001; Kuo 2003; Wolf 2003; Chappell 2009).

As an emerging field in the natural sciences, urban ecology has yet to be delineated by a concrete definition of concepts and theory, even of essential terms such as “urban” (McIntyre, Knowles-Yanez, and Hope 2000; Grimm and Redman 2004). McDonnell and Pickett describe the process of urbanization as “an increase in human habitation, coupled with increased per capita energy consumption and extensive modification of the landscape, creating a system that does not depend principally on local natural resources to exist (1990).” Indeed, the urban area can be considered its own unique ecosystem, with an environmentally-selected group of organisms that characterize the internal habitats (Odum 1969). Often these urban areas share a similar flora and fauna that is characterized better by its degree or period of urbanization than other regional factors (McKinney 2006, 2008; Clemants and Moore 2003; La Sorte and McKinney 2006). Some use human population density figures to define the urban setting, following the US Census Bureau; others consider the percentage of impermeable land cover (McIntyre, Knowles-Yanez, and Hope 2000).

Detroit presents a unique set of characteristics that make it an important addition to urban ecological studies in North America (see Preface). Detroit was first a French agricultural village, which grew to be a major shipping port in the late 1800s, then experienced the transition into industrialism and subsequent continuing decline. The varied history of the land use in cities may offer insights into the long-term processes of urbanization. For instance, the urban core is generally considered the least vegetated, most altered area, with irregular rings of development that diminish in population density as one moves away from the center (Dickinson 1966 in; Odum 1969). However, this concept is too simplified, and further research has generally found that as neighborhoods increase in socioeconomic status, so does the diversity and abundance of plants and birds (Kinzig et al. 2005; Hope et al. 2003; Martin, Warren, and Kinzig 2003). Yet, due to the dramatic influence of vacancy in Detroit, the opposite trend was found in the city – vegetation is increasing, and is most abundant in the least affluent areas of the city (Ryznar 1998). The American Forests survey of the City of Detroit found that the city’s land, encompassing 370 square kilometers, is covered by 46.90% impermeable surfaces, 31.23% shielded by tree canopy, 20.02% open space, including ruderal vacant lots, 1.50% bare soil and 0.35% water (2006). Addressing the ecology of Detroit at this point in redevelopment may help urban planners and land managers to evaluate revitalization projects, such as the current

construction of greenways and walkways, in both traditional socioeconomic terms and with a bio-ecological perspective.

The following chapters present the results of a study of beetle biodiversity in Detroit. The study aimed to describe the beetle communities occupying three greenspace habitats – remnant forests, community gardens, and vacant lots – and discover similarities and differences in the beetle assemblages. These twelve sites, including four replicates of each habitat type, were considered “urban” land uses by the fact that they fell within the city limits of Detroit. The remnant forest fragments were chosen for their century-and-older tree stands and relatively native flora and are most reflective of the original landscape of Detroit pre-European settlement and urban growth. The post-urban ruderal lots were all formerly residential or non-industrial commercial properties that have been vacant of structures for at least 15 years. This land use was chosen because it is the most common type of vacancy that occurs in the city, considering the loss of over half of the former 1.8 million residents, and is therefore one of the greatest challenges in renewing the city. Furthermore, residential lots are less likely to have confounding effects from industrial soil pollution, and are likely to display the pattern of re-vegetation reported by Ryznar (1998). Community gardens, representative of a movement in Detroit and throughout the world to address blight and food insecurity through urban agriculture (Dubbeling and Santandreu 2003; Quito Declaration 2000), are maintained an early successional regime of volunteer forbs due to annual tilling, along with intentionally planted agricultural plants.

In addition to internal floristic surveys, a suite of habitat variables, including the presence of impermeable ground cover, was used to characterize the land surrounding each of the study sites in order to better define the immediate external surroundings. I utilized pitfall trapping to capture invertebrates (Ausden 1996) within each site. Though much of the vertebrate fauna that occupied the Detroit area has disappeared¹, arthropods remain and can reflect essential elements of urban biodiversity. Arthropods such as beetles are good candidates for study because they 1) are hyper diverse and therefore may represent the overall biodiversity of an area; 2) have

¹Father Louis Hennepin, a member of LaSalle’s exploratory crew in the late 1600s, recorded that “The country is stocked with stags, wild goats and bears...” (Gavrilovich & McGraw 2001, p. 30); and several of Cadillac’s early reports of the area note “luxuriant grass upon which fatten woolly buffaloes” and owl including turkeys, golden pheasants, quail, partridge, woodcock, and doves (Farmer 1969, p. 11). Though early explorers were prone to hyperbole, the use of the land by northern Iroquois, Potawatomi, Ottawa, and other Native American Indian tribes for winter hunting grounds indicates that larger game was once present in the area (Gavrilovich & McGraw 2001, p. 32, 91, and 131).

relatively short generation times that can respond to anthropogenic changes; 3) are relatively easy to sample; 4) represent several functional groups; and 5) have other important social, agricultural and economic roles in human-modified environments (McIntyre 2000). A few studies exploring arthropod communities in association with urban change or land use have been conducted in association with the Long Term Ecological Research (LTER) site in Phoenix, Arizona (McIntyre and Hostetler 2001; McIntyre et al. 2001; Cook and Faeth 2006).

A group of scientists have created “GLOBENET”, an organization that studies carabid beetles (Coleoptera: Carabidae) in forests along urban-rural gradients to evaluate their potential use as bioindicators organisms in forest settings (Kotze and Niemelä 1998-present; Rainio and Niemelä 2003). Numerous urban-rural gradient studies (Reale and Blair 2007; Elek and Lovei 2007; Magura, Tothmeresz, and Lovei 2006; Magura, Lovei, and Tothmeresz 2008; Weller and Ganzhorn 2004; Niemelä and Kotze 2000) have been completed in European countries, and associated studies were conducted in Canada and Japan (Ishitani, Kotze, and Niemelä 2003; Niemelä et al. 2002); however, no such urban-targeted study appears to have been completed in the United States. General findings of these studies point to little if any decline in carabid species richness based on the proximity of the forests to the urban center. However, carabid assemblages do appear to decline in body size with the degree of adjacent urbanization, and the number of flightless species also declines with degree of isolation. Other studies of carabid populations in response to agriculture and forest disturbance have been conducted in Michigan (Moghtader 2004; Petrillo 2006), which may offer some opportunity for comparison.

This thesis includes an extensive preface, which provides some historical, social and economic background about Detroit, the urban setting for this study. As urban ecological research is an effort to understand a system that is inherently coupled with human actions (Liu et al. 2007), it is valuable to understand the history and current condition of the human population. The racial stratification of the city (Trowbridge 2002) and the current social issues related to poverty (U.S. Department of Labor 2008), lack of access to healthy food (Mari Gallagher Research and Consulting Group 2007; Pothukuchi and Thomas 2004) and important community amenities may draw parallels to other international developing urban areas. I thought it important to include this information as an accompaniment to the ecological study because a lack of social context of this type can prevent urban ecological studies from being compared between cities or utilized future in multi-disciplinary studies (McIntyre, Knowles-Yanez, and Hope 2000).

Chapter II is an analysis of the carabid beetle assemblages captured and their relationship to the habitat types. This study seeks to identify differences in biodiversity measures and morpho-ecological groupings of carabid beetles, as well as the change in the capture rate over the season. An attempt to relate the specific habitat characteristics to the biodiversity measures in respect to habitat type is also presented. The value of different land-use types for conservation of native fauna is also considered. This information may be valuable to land managers who are interested in having a relatively simple method for environmental sampling that may reflect the response of fauna to the habitat, and adds to the base of knowledge on carabid beetles that may be used for identifying more universal characteristics of the family for biological monitoring.

Chapter III explores three different taxonomic groupings of beetles: all the morphospecies captured, all the families of beetles captured, and the carabid and rove beetles, which are most appropriately considered by the pitfall trap methodology. The biodiversity measures and seasonality of these groupings is considered in order to better understand the differences in response to the three land use types. Comparing the cluster analyses of the three beetles groupings highlights the fact that different ecological phenomenon are recognized by selecting different taxonomic groupings of beetles. By widening the scope of fauna considered, this paper is intended to present a greater breadth of information about the sites, and provide information relevant to fields other than natural area conservation.

In the concluding Chapter IV, I summarize the results of the study, and explore opportunities for future formal and informal investigations of urban ecology and agroecology in Detroit. Though professionals in conservation fields generally employ monitoring programs, a “citizen science” monitoring effort could contribute to the data available to academic researchers, and would complement many ongoing efforts to supplement education and reconnect with nature in Detroit.

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CHAPTER II

CARABID BEETLES OF DETROIT GREENSPACES AS AN INDICATOR TAXON

As the human population continues shift to urban areas, it is essential that science provide a better ecological understanding of the urban environment for the informed management of urban areas and more sustainable urban development (McDonnell 1997; McIntyre et al. 2001). Urban areas can be defined in many ways, but are generally characterized by a heterogeneous landscape structure including greater impermeable cover, pollution of all types, an altered local climate, significant anthropogenic alteration of natural resources and significantly increased energy use (McIntyre 2000; McIntyre, Knowles-Yanez, and Hope 2000).

Urbanization can be a “biological homogenizer,” encouraging species adapted to the urban environment (McKinney 2006). However, this is not synonymous with loss of biological diversity; species introductions and the concentrated food resources of urban areas foster species diversity that can exceed that of the preexisting “natural” environment (McKinney 2006). Novel ecotypes may occur, with different ecological pressures and feedback loops that influence the community of organisms (Odum 1969).

Arthropods are prime candidates for urban studies, as they 1) are hyper diverse and therefore may represent the overall biodiversity of an area; 2) have relatively short generation times that can respond to anthropogenic changes; 3) are relatively easy and uncontroversial for sampling; 4) represent several functional groups; and 5) have other important social correlations with human-altered habitats, such as economic impacts on urban forests or indicators of organic pollution in waterways (McIntyre 2000). While sampling the arthropod fauna of an urban area is likely to yield interesting and important biological information, the capture and identification of such a range of species is too time and resource consuming for ecological monitoring.

The use of arthropod indicator taxa is a commonly applied ecological method, but one which needs to be tailored to the environment of interest (e.g. Kremen 1994; Parisi et al. 2005; Angold et al. 2006). Carabid beetles (Coleoptera: Carabidae; commonly known

as ground beetles) are a potential indicator taxon within terrestrial forest environments (Rainio and Niemelä 2003). Relative to other arthropod groups, carabid beetles are relatively large and economical to capture, are sensitive to habitat changes, and have a well-documented taxonomy (Thiele 1977; Lovei and Sunderland 1996). The “Global network for monitoring landscape change”, or GLOBENET, is a research group created to study carabid beetle diversity along urban-rural gradient in countries across the world with the intention of producing a monitoring program for forest system management based on carabids (Niemelä and Kotze 2000). GLOBENET’s standardized method of pitfall trapping of carabid beetles is a commonly utilized method for obtaining a qualitative inventory and comparison of species assemblages in temperate forests (Spence and Niemelä 1994; Luff 1975; Niemelä and Kotze 2000).

Most GLOBENET studies to date have been conducted in Europe, and show that poorly dispersing specialist species tend to decrease with increased disturbance, while small generalist species with good dispersal ability tend to increase (Howe and Enggaard 2006; Magura, Tothmeresz, and Lovei 2006). Similar to plant studies (Clemants and Moore 2003), carabid studies have also found that the species diversity is usually maintained or declines insignificantly in urban forests, but species composition changes, with several exotic generalist species replacing native specialist species. For instance, the GLOBENET studies conducted in Canada found that 77% of carabid abundance consisted of European generalist species (Spence 1998). As carabids are a well-studied group in Michigan (e.g. Petrillo 2006; Moghtader 2004; Werner and Raffa 2000), reference collections and identification materials are available and could be useful for a variety of ecological studies that could contribute to monitoring or adaptive management of forests and urban natural areas in Michigan.

Though carabids are considered useful bioindicators, there is not enough data on their relationship with other plant and animal species to consider them a surrogate for other species or representative of the biodiversity of the area of interest (Rainio and Niemelä 2003; Lindenmayer, Margules, and Botkin 2000). For instance, examining the taxonomic species diversity of carabids alone has been shown to be a poor indicator of habitat type, while the morphological characteristics and differences in carabid assemblages are often associated with habitat (Deichsel 2006; Gobbi and Fontaneto

2008; McIntyre 2000). Studies have revealed relationships between plant structure and carabid microhabitat preferences (Brose 2003), while landscape-level characteristics can assist in determining the overall biodiversity of the site (Lindenmayer, Margules, and Botkin 2000; Brose 2003; Deichsel 2006). The use of historical insect collections may provide some in situ information related to change in the composition of Carabid communities and morphological characteristics of species over time, but collector bias appears to render species richness estimates inaccurate (Petersen, Meier, and Larsen 2003).

As ecological researchers move away from the equilibrium perspective that espouses the idea of a singular climax community, and towards the “nonequilibrium paradigm” of community formation, the role of disturbance becomes a central element in the environment (McDonnell 1997), and urban environments a prime setting for researching these interactions. Human disturbance may induce selection pressures that are of interest on the population level, and urbanization presents a framework for the study of such changes in morphology as well as the extinction dynamics of metapopulations (Frankie and Ehler 1978). Urban areas can be considered a unique ecosystem (Parlange 1998; Odum 1969) and to understand the effects of urbanization on biodiversity, it is essential to also consider the matrix of land uses beyond remnant forests that remain after land conversion (Vandermeer and Perfecto 2007; Gaublomme et al. 2008).

Fostered by efforts such as the Long Term Ecological Research Site in the city of Phoenix (Grimm and Redman 2004; Hope, Grimm, and Redman 1998), ecological studies considering the changes in arthropod diversity and composition in urban areas, before and after urbanization [i.e. clearing land for urban development] are emerging in the literature of urban ecology in North America (e.g. McIntyre et al. 2001). Few studies, if any, exist that consider the post-development landscape of declining urban areas, such as Detroit.

Detroit offers a unique opportunity to study the ruderal communities of the relatively unmanaged regrowth of vegetation after half the human population has vacated the residential and industrial areas of the urban center (Bureau 2006; Gibson 1998; Ryznar 1998). The relative lack of food and economic resources for the human

population (Brookings Institution et al. 2006; Mari Gallagher Research and Consulting Group 2007; Pothukuchi and Thomas 2004) has resulted in a burgeoning urban agriculture effort (Gopakumar 2005; Hanna and Oh 2000), including hundreds of community and pocket gardens. These new green areas offer another distinctive land use that may foster unexpected diversity. Finally, amid the industrial legacy of the city are a few high-quality remnant forests, which may be the only vestiges from the wetland forests that once occupied the area (Weatherbee and Klatt 2004; Comer et al. 1995).

Understanding the effects of various land uses and related disturbance regimes on urban biodiversity is valuable in the consideration of creating ecologically and socially beneficial urban landscapes (Parlange 1998; Gyllin 2004). Studies of indicator species such as carabid beetles could be useful information for prioritizing management goals, defining restoration goals, evaluating biocontrol potential, or identifying pollution areas. This exploratory research aims to answer the following questions: 1) Do differences in species richness, activity-density, and species composition occur between vacant lot, community garden, and remnant forest habitat types? 2) Do these habitat types foster carabid communities with unique species assemblages or morpho-ecological characteristics? 3) What habitat factors might contribute to carabid assemblage differences?

Methods

Site Selection

The study was conducted during the summer of 2007 in Detroit, Michigan (42°19'53.76"N, 83°2'51"W, approximately 183m in elevation). Greenspace land uses were represented as four replicates each of urban forests, vacant lots, and gardens for a total of twelve sites (Figure 1). Table 1 provides a description of each site. Habitat characteristics for each site, including vegetation, land cover quantification are provided in Table 2, and weather during sampling periods is provided in Table 3, as recommended by McIntyre (2000) (habitat characteristics of each site are recorded in Appendix A). Remnant forests sites were selected with the assistance of former city naturalist Susan Campbell, based on the age of the tree stands, floristic quality, and suitability for study (June 2006). Ms. Campbell also provided recent floristic inventories, which were used to

evaluate the Floristic Quality Indices, a ranking system for the conservation value of the natural areas (see Table 4 for further information). The higher the FQI, the more important the natural area is considered in terms of conservation value. The rank of these four forests, from lowest to highest in FQI, is Balduck Park, Belle Isle, Palmer Park, and River Rouge Park.

Vacant lots were identified as former residential or non-industrial commercial use parcels; each lot was confirmed as to its former use through exploration of former tax records and zoning maps (City of Detroit Finance Department 2006; City of Detroit Planning and Development Department 2006). Historical aerial photographs (USGS and USDA 1949-1997) and Sanborn Fire Insurance Maps (1992) were consulted to ascertain that the vacant lots were free of a structure for at least fifteen years prior to the study; this time frame appears to allow the plant and insect communities respond to the disturbance of structural clearing and respond to continuous mowing (Gilbert 1989). Additionally, illegal dumping observed in photographs would lead to the exclusion of a site due to more probable soil contamination. Urban gardens were included on a volunteer basis, in response to a request made to participants in the Detroit Agriculture Network. Each garden was managed without chemical pesticide use.

Carabid Sampling

Six ground pitfall traps consisting of two nested 16 oz. plastic tubs [11.4 cm in diameter; 7.6 cm in depth] were placed in each site along two short transects with three traps in each transect. Traps were 5 meters apart, and the two parallel transects were 5 meters apart, forming a rectangle (Figure 2 and Figure 3). The cups were sealed with a lid when not in use. This diameter of trap is unlikely to capture rare species or largest of invertebrate species, but acceptably characterize the dominant fauna active within the ground layer as well as larger, more inclusive traps without the mammalian and amphibian by-catch (Work et al. 2002).

The traps were open from 15-18 May, 16-19 June, 16-19 July, and 15-18 August of 2007, filled with approximately 200 ml saturated saline solution with a minute amount of unscented detergent to break surface tension. Saline solution is a preferred neutral field sampling preservative, with little attractive or repulsive characteristics for

Carabidae, while preserving important morphological components of the insects (Perner and Schueler 2004; Sasakawa 2007). A plastic plate with three nails for “legs” was placed as a roof approximately 2-3 inches above each open trap to exclude rainwater and non-target taxa.

Though some trap-to-trap interference is likely (Digweed, Currie et al. 1995), the plot size was kept similar across sites in order to accommodate the smallest of sites, G 4. The forest traps were placed in areas away from the forest edges to avoid edge effects, and at relatively higher elevations within the site, as the forest sites were seasonally flooded during the first sampling date in May and the water table would otherwise push the cups from the ground. Garden traps were placed near or among row plantings, but on request of the gardeners care was taken to choose a layout that would least disturb cultivated plants. All gardens were tilled approximately one to two days prior to the trap installation. Vacant lot traps were placed towards the center of the ruderal vacant area to avoid edge effects, though an attempt was made to place traps near occasional vegetation, such as trees or shrubs, in order to detect carabid species with plant structure preferences that may be present (Brose 2003).

Specimen cups were retrieved from the pitfall trap and the contents was filtered with deionized water to remove salt solution, and then stored in a 70% ethyl alcohol solution in refrigeration. Carabid beetles were first sorted by morphospecies and representative individuals were pinned. Peter W. Messer, nearctic carabid taxonomist of Wisconsin, identified each representative individual to the species level. Information regarding the pitfall station, site, habitat type, and sampling date was maintained with each individual.

Habitat Characterization

For each site replicate, habitat characteristics were measured at three spatial scales (1 ha, 400 m², and 1 m²) to examine the type and scale of factors that may correlate with carabid diversity and abundance. At each site, a 100 x 100 m plot (1 ha) and a 20 x 20 m plot (400 m²) was established, centered on the middle the baiting areas. In each 100 x 100 m plot the percent of ground covered by a) bare ground, b) understory vegetation (e.g.

forbs, grasses, etc.), c) shrubs, d) concrete, and e) buildings was estimated. The number of trees >30 cm circumference at breast height (cbh) was also counted.

Data on trees, shrubs and canopy cover was collected within 20 x 20 m plots. For trees a) counted and identified all trees >30 cm cbh, b) measured circumference at 1.37 m above the ground, and c) estimated height. For shrubs and tree seedlings, we a) counted and identified all stems < 1 m height, b) measured circumference at the ground, and c) measured height. The total richness of woody plants at each site was calculated by combining tree and shrub data. The percent canopy cover for each 20 x 20 m plot was estimated with a concave vertical densiometer. We took densiometer readings at the center and the four cardinal corners of the plot at the start of each pitfall sampling, and calculated mean values for each site over the season.

To establish the 1 m² plots, the 20 x 20 m plot were divided into 4 sections of 5 x 20 m. Within each section we randomly placed a 1 x 1 m plot using a generated random number chart to measure percent covered by a) bare ground, b) grasses, c) forbs/herbs, d) rocks/wood panels, e) leaf litter, and f) fallen branches. For each 1 x 1 m plot, we also recorded a) the height of the tallest non-woody plant, b) number of individuals of forbs/herbs, and c) number of species of forbs/herbs and grasses. The habitat data were collected once at each site between May-September 2007.

Morpho-Ecological Traits

After the identification of the carabid beetles, each species was investigated using *A Natural History of the Ground Beetles (Coleoptera: Carabidae) of America north of Mexico* (Larochelle and Lariviere 2003) to determine traits that may provide insight to habitat interactions. The average length (a proxy for size), and the flight ability was determined for each carabid species. Though the eating guild is often considered in such studies, the information was indeterminate in many cases, and the catholic eating habits of the carabid beetles means that the natural feeding habits of the beetles may adapt over habitats and seasons. Therefore eating guilds were not included in this analysis.

Historical Carabid Collection

The biogeographic origin of each species was determined with Laroche and Lariviere (2003). Species were identified as either native to North America or introduced species, and in the event of an exotic species, the place of origin was recorded.

The collection of carabid beetles at the University of Michigan Exhibit Museum of Natural History was reviewed during the summer of 2008. Each specimen that was labeled as collected in Wayne County, Detroit, or a park or feature that is known to be in Detroit (i.e. Palmer Park or Belle Isle) was recorded. All information on collection date, location information, and other detail such as collection method provided on the specimen label was recorded.

Analysis

EstimateS software was used to create species rarefaction curves, estimate species richness, and to calculate the shared species indexes (Colwell 2005). PAST software was utilized for cluster analysis (Hammer et al. 1995-2008). All other statistical analyses were conducted with SPSS 17.0 or 16.0 (SPSS Inc. 2008).

Activity Density and Seasonality of Carabid Capture

As the capture rate of carabid beetles in pitfall traps is associated with their activity level and local density, carabid abundance is referred to as “activity-density” (Thiele 1977). The activity-density is calculated as the total number of individuals captured at the site divided by the total number of traps. As traps may have interactions within the same site (Digweed et al. 1995), and most sites are heterogeneous at the microsite scale, it is appropriate to consider each trap as a subsample of the site area (Greenslade 1964; Spence and Niemelä 1994). Here, the activity-density was considered the mean number of Carabidae beetles captured per trap per three-day sampling period. Activity-density data was also pooled for the four-month sampling period at each site for the diversity analyses.

All activity-density measurements were transformed prior to analysis using the SQRT ($x + .05$) transformation to normalize count data that follows a Poisson distribution and to equalize variance among the sites. The one-way repeated-measures General Linear Model with Tukey’s Honestly Significant Differences (HSD) post hoc

test was run to determine activity-density differences between monthly samplings and between habitat types. The values from the Wilks' Lambda multivariate test are reported, which determines if significant differences exist between the months. The Greenhouse-Geisser method is reported for comparisons among sites of the same habitat, due to the lack of sphericity (or presence of correlation) in the data. The Tukey's HSD corrects for the uneven sample number due to the missing May data for forest site 1. A one-way ANOVA with Tukey's HSD was used to compare the means of the pooled seasonal activity-density by habitat type.

Species Richness

Rarefaction curves for each site and habitat type were created using EstimateS software. Plotting the 95% confidence intervals for the observed species curves allowed for the determination of significant differences between sites and habitats. The Jackknife1 (Burnham and Overton 1979) non-parametric approach, which uses only the number of species observed and the number of species occurring only once to evaluate the number of unseen species (Chao 2005), was used to estimate the number of species present at each site. For diverse taxa, the Jackknife1 estimator performs with less bias and greater precision than other non-parametric estimators (Colwell and Coddington 1994).

Actual species richness differences between habitat types were considered by graphing the Jackknife1 extrapolated species accumulation curve with the calculated 95% standard deviation. A Tukey's HSD post hoc following a one-way ANOVA using the 1) Jackknife1 species richness estimates and 2) observed species for each site was performed to confirm the visual observations of the differences between habitat types plotted from the data in EstimateS.

Cluster Analysis of Shared Species

Hierarchical cluster analysis using Morista's index for abundance data (Morista 1959) based on the unweighted pair-group average (UPGMA) linking method was used to determine the relationships within site and habitat types using PAST v. 1.88. A one-way analysis of similarity (ANOSIM) was also conducted using PAST v. 1.88 to determine significant differences between habitats based on a non-parametric test of the

Morista Index values. The Morista Index for between-site and between-habitat comparisons of species richness was computed using EstimateS. The index approaches 1 when sites are more similar, while a score of zero indicates no shared species. This analysis is utilized in other studies of arthropod response to habitat heterogeneity (McIntyre et al. 2001; Petrillo 2006).

Habitat Characteristic Comparisons

The habitat characteristics were compared to 1) carabid activity-density and 2) Jackknife1 species diversity estimates using a series of backwards elimination, or subtractive, multiple regressions. Though backwards multiple regression model is prone to large errors due to collinearity or redundancy of the predictor variables (the habitat characteristics in this case), the backwards elimination method was utilized in this study to capture possible variables of interest for further research. All the predictors were centered by standardization (Z-score in SPSS) before running the regression, which assists in decreasing the effects of collinearity among the variables.

The variables were separated by the sampling scheme at the 1m², 400m², and 1 hectare scale; variables were included in the model at the 0.05 level, and excluded at the 0.10 level of significance. For the 1 hectare scale regressions, the variable “number of buildings” was excluded, as it was considered possibly redundant to the building basal area. The “site area” variable, a measure of the contiguous area considered part of the habitat type at each site, was also compared to the carabid biodiversity measures using a single linear regression model.

Additionally, separate backward regressions were conducted for the forests sites alone, as they differed from the other habitat types in the total species richness and activity-density measures and were significantly different in internal and external structure compared to the other two habitats (see Chapter 3). Selection of factors that have been shown to influence the carabid species richness in forest areas, including the patch size, percent cover of downed wood, percent cover of leaf litter, and site age were included in the forest regressions.

Morpho-ecological and Historical Traits

The two categories of morpho-ecological traits, length and flight, were separated into categories. Based on the flight capability, carabid species were separated into the following groups: 1) Macropterous and submacropterous (frequent or occasional flyer); 2) dimorphoc, indicating that flight ability tends to be lost over time in populations in a stable environment or conversely maintained in unstable environments; and 3) brachypterous (beetles with no flight capability) (Gobbi and Fontaneto 2008; Deichsel 2006). The average length of the beetle species were classified as either 1) small, with a length less than 7mm; 2) medium, with a length less than or equal to 7mm and less than 15mm; or 3) large, with a length of 15mm or greater (Cole et al. 2002).

For each site and habitat, the percent of species in each of the above categories was computed. As pitfall trapping is likely a poor measure for actual density in areas (Spence and Niemelä 1994), these percentages were not weighted by the individual counts of each species.

Results

Activity Density

The overall activity-density of the vacant lots was consistently lower than that of the other habitats, while the gardens and forests maintained a relatively higher activity-density that did not differ significantly from each another (Figure 4). The overall activity-density significantly differed between the habitats ($F = 11.659$, $df = 2$, $p = 0.003$), with significant differences between the forests and vacant lots ($p = 0.003$), and vacant lots and gardens ($p = 0.024$) indicated in post hoc tests. However, no significant difference was found between the forests and gardens for the pooled data for the season.

The activity-density significantly varied with the month of sampling ($F = 19.919$, $df = 3$, $p = 0.002$), and significant interactions between month and habitat exist ($F = 5.875$, $df = 6$, $p = 0.005$). In May, the forests had a low average activity-density similar to that of the other habitats, but maintained a higher activity-density in the other months (Figure 4).

The highest combined activity-density occurred in June, with another peak in August. A drop in the garden activity-density in July appears to be the primary factor in this dip in overall activity-density during this month. The number of carabids in the vacant lots showed a small but steady increase throughout the season, but the total vacant lot

activity-density is only a little less than a sixth of that found in the gardens and an eighth of that found in the forests.

The within-habitat type comparisons of activity-density by month also determined that there were significant differences among the sites ($F = 5.417$, $df = 1.37$, $p = 0.032$). The activity-density at some sites had a comparatively high activity-density in comparison with the other sites. There was high within-habitat variability among garden and forest sites.

Species Richness

The rarefaction curve of accumulated species (Figure 5a.) suggests that there is little difference in species richness between the habitats, though some separation in the habitat curves may be emerging as more individuals were encountered, as in the gardens and the forest. When the curves were truncated to compare species richness among habitats with the same number of individuals (Figure 5b.) it becomes clear that there are no significant differences among habitats.

Examination of the estimated species diversity (Figure 6a.) suggests that greater carabid diversity exists in the forests in comparison to the gardens. A closer examination of the species estimate curves (Figure 6b.), the vacant lot habitats appear to be leveling off, and the trend of this curve suggests that the maximum number of species will be less than the number of species of carabids that are found in either the gardens or the forests. The results of the ANOVA comparing the observed and estimated species diversity for each site did not indicate a significant difference in species diversity between the habitats.

Some trends appear when the average number of species found in each habitat is considered (Figure 7). In both the forests and the vacant lots, the number of carabid species found appears to increase over time, while in the number of species in the gardens appears to remain steady, with small vacillations over the season such as a decrease in July. Overall, eighteen species were captured in forests samples, compared with eleven species found in the gardens and six species in the vacant lots (see Appendix B for a list of the carabid species). Even when averaged across the season, the forests samples contained in one-third more species than the gardens, and two-thirds more than that found on vacant lots (Figure 7). When the total number of species captured at each site is

considered, F1 [Belle Isle] is prominent. Eleven species were found at this site, which is approximately twice that found at any other site, even with a loss of all the May samples at F1 due to flooding and animal disturbance.

Habitat Associations

Using the backwards regression method of analysis, several of the habitat factors appear to be correlated with activity-density and species diversity, though the factors that affect these two measures of diversity appear to be different. It is important to note that based on a review of the correlation matrix and collinearity diagnostics, many of the habitat characteristics are significantly or highly significantly correlated to one another (dependent). This indicates that some of the correlations may be spurious and the significance of the following results should be considered preliminary.

The area of the habitat patches was not found to be a significant driver of carabid activity-density. At the 1 hectare scale, the most significant predictor is the percent ground cover vegetation, which is negatively correlated with activity-density and outperforms all other predictors when considered alone ($R^2 = 0.535$, $df=1,10$, $F=11.497$, $p = 0.007$). The percent of bare ground, percent of vegetation cover, and percent of shrubs all contributed to a statistically significant model at the one hectare scale ($R^2 = 0.636$, $df = 3, 8$, $F = 4.655$, $p = 0.036$). However, these factors are highly correlated and therefore the single factor of percent vegetation covering the area would seem an acceptable single measure.

At the 400 m² scale, percent canopy cover was a marginally significant positive predictor for carabid activity-density ($R^2 = 0.329$, $df = 1, 10$, $F = 4.914$, $p = 0.051$). At the 1 m² scale, including all the variables yielded a significant result ($R^2 = 0.964$, $df = 8,3$, $F = 10.173$, $p = 0.041$), which may reflect the high degree of correlation between the predictors. The percent cover of rocks, percent cover of downed wood, percent cover of forbs, percent cover of bare ground, percent cover of leaf litter, and percent cover of grass all contribute to the most significant regression model ($R^2 = 0.957$, $df = 6, 5$, $F = 18.730$, $p = 0.003$). The percent grass cover appears to be the most highly correlated when all the prior factors are compared to the activity-density in a single linear regression ($R^2 = 0.7137$), with a negative relationship to activity density.

The second diversity factor, carabid species richness, was also compared to the same habitat characteristics. At the 1 ha scale, the percent of bare ground was the most significant predictor ($R^2 = 0.435$, $df = 1, 10$, $F = 7.702$, $p = 0.020$). The percent of building area with percent building area was also significant ($R^2 = 0.503$, $df = 2, 9$, $F = 4.558$, $p = 0.043$). At the 400 m² scale, no habitat characteristics produced a correlation at the 0.05 significance level, but the average tree height, average percent canopy cover, and average tree circumference meet a less stringent 0.10 significance ($R^2 = 0.528$, $df = 3, 8$, $F = 2.986$, $p = 0.096$). At the 1 m² scale, no variable produced a significant association, although the percent cover of rock and wood panels, average of percent cover of forbs and herbs, and the average of percent cover of grasses came near significance ($R^2 = 0.570$, $df = 3, 8$, $F = 3.528$, $p = 0.068$).

The tolerance value was reached in the regression models of the forest sites in both the selective variable model and for all variables in general. This indicates that many of the factors are highly collinear, meaning that they are highly correlated, and therefore are unlikely result in accurate correlations.

Shared Species and Cluster Analysis

Even though the different habitats have a few shared species, the ANOSIM based on Morista Index values confirmed that the forests had significantly different species composition from the gardens ($R = 0.8229$, $p = 0.0276$) and the vacant lots ($R = 0.526$, $p = 0.0253$) based on the observed species diversity and abundance. An examination of the Morista Index (MI) values (Table 6) and cluster analysis (Figure 8) shows that the gardens and vacant lots are more similar in species composition ($MI = 0.436$) to one another than the forests are to either habitat. The forest was the most distinct habitat, sharing only one species with the gardens ($MI = 0.001$) and one species with the vacant lots ($MI = 0.022$).

At the site level, G1, G2 and G3 were grouped very closely and have MI values near one, indicating very similar carabid species assemblages. V3 and V4 were compositionally more similar to these three gardens than to G4, which branches very early from these garden and vacant lot sites. The MI values show a slight similarity between G3 and G4, but no similarity between G4 and G1 or G2. There are few shared

species between most of the vacant lots themselves, with the exception of V3 and V4 (MI = 0.897). Of the remaining vacant lots, V1 is distantly similar to the forests while V2 is on an entirely independent branch, illustrating that the species found in the lot was not found at any other site. The forest areas and V1 occupy a separate initial branch of the Figure 9, indicating that they share no carabid species with the other sites. The forest sites have carabid assemblages that overlap to varying degrees, indicated by intermediate levels of branching and MI values.

Morpho-ecological and Historical traits

The percent of species that occupy each of the morpho-ecological categories is presented in Table 5. Most carabids collected were medium sized; the larger carabids were primarily found in forests and gardens. F2 had 4 out of 5 (80%) of the large carabid species in its assemblage. Most captured species were flying species; the greatest percentage of flightless species (60%, or 3 out of 5) also occurred at F2. Native species accounted for about one third of species in the gardens. Two of the forest sites, F3 and F2, had 60-70% native species, while 100% of the F1 species and 83.33% of the F4 species were native. When considered at the habitat level, the highest percentage of native species occurred in vacant lots, with 83.3% of the species in this habitat originating in North America.

According to specimen data from the University of Michigan Exhibit Museum of Natural History in Ann Arbor, Michigan 158 species of carabid beetles occurred in the Detroit region in the early 1900's. Of these species, only 13 were collected with pitfall traps during this study. The garden sites had nearly twice the proportion of species with a representative in the historical collection than the other two habitats. The information from the museum is provided in Appendix C.

Discussion

Activity-Density and Seasonality

Significant differences in activity-density occurred between vacant lot, community garden, and remnant forest habitat types in regard to season. The low activity-density of forests in May is likely due to the effects of seasonal flooding or a

later seasonal emergence of the carabids inhabiting these environments. Like other arthropods, the amount of carabid movement is affected by weather and temperature (Greenslade 1964). The activity-density was notably lower in sampling periods that had greater precipitation (Table 3). The fewest carabids were captured in May (Figure 4), when a total of 1.56 cm of rain fell and the average temperature was about 10 degrees cooler than the other sampling periods (13.5°C, compared to 24.25°C, 23°C, and 21.75°C, respectively). The lessened activity-density in July may be due to the 1.02 cm of rain that fell over the sampling period, or may be a phenomenon related to a gap between the emergence of overwintering of beetles and emergence of the a second generation. Interestingly, it appears that the higher June and August activity-density for the gardens is generally dependent on G2 and G4, respectively, which may be one of the factors contributing to the significant differences among sites in the same habitat. The forests appear to have more equally distributed activity-densities, though F3 has higher activity densities in all months but August. F3 [River Rouge Park] was also the largest park, and the sampling site was located near a river, so this may have positively influenced the capture rate due to greater resources.

For the sake of comparison with the GLOBENET rural-urban “treatments”, the habitat types used in this study could be classified on based on a disturbance gradient, with rural land to be the least disturbed and urban land to be the most disturbed. The remnant forests are the least disturbed, followed by the vacant lots which are mowed approximately three times a season by the City of Detroit making the mix of herbaceous, grass and woody species similar to that in suburban areas, and finally the gardens, which are frequently disturbed by tilling and planting of exotic species. In this study, the greatest activity-density and biodiversity were found in the relatively undisturbed forest areas and the least in the moderately disturbed vacant areas. The gardens areas contained a great enough activity-density as to not be significantly different from the forests. This does not correlate with the findings of most rural-urban studies that tend to find fewer carabids in the most disturbed areas (Niemelä et al. 2002; Ishitani, Kotze, and Niemelä 2003). I presume that this is because the increased complexity of the environment: a diversity of nutrient-rich plants, improved soil and water access, and a resultant increase in prey provide resources for the carabids, while the relatively open ground-level habitat

allows for more successful hunting. Due to the addition of these resources, the gardens may actually be more akin to a moderately disturbed suburban area with diverse, well-maintained plantings. The consistently low levels of activity-density in the vacant lots may be a slight underestimate of the carabid assemblages living there because dense grass can inhibit beetles from the ground-level travel that results in their capture in pitfall traps (Greenslade 1964). Similar to other gradient studies, the observed species richness does not significantly differ between the habitat categories, though the composition of the carabid communities is quite different (Niemelä et al. 2002; Howe and Enggaard 2006).

Morpho-ecological Interactions

The highest activity-density and species richness were found in the forest habitats, and among these, F1 had about twice the species richness of any other site. (Although this difference was not significant, the curves for richness estimates suggest that with a higher sample size, the observed differences may become statistically significant.) All species found are of North American origin, and one, *Pterostichus praetermissus*, represents a state record. Though this was the first recorded occurrence of the species in Michigan, others have been identified *P. praetermissus* in nearby Ontario, Canada (Goulet and Bousquet 2004). It is similar in appearance to *Pterostichus commutabilis* and was only recently re-differentiated; this species is not considered an exotic to North American, and its discovery is likely a result of either sampling effort or a short migration from across the Detroit River to F1 [Belle Isle] (Messer 2008). These findings suggest that despite the low Floristic Quality Index value (Table 4), F1 may represent a unique high-quality environment for carabid beetles. One explanation for the high number of native species is that the F1 forest exists on an island connected to the mainland of Detroit by a bridge that harbors no vegetation, and therefore it may have been relatively protected from the introduction of exotic carabid species that were found in the other forests.

The carabid assemblage of F3 suggests that not all forests foster similar biological diversity. Though the activity-density at F3 was greater than at any other site, and the size of the habitat was much larger than that of the other parks, only five species were found. Of these species, two were exotic, and four of the five species were considered

large. This high activity-density and body size may indicate a resource-rich environment; many slugs were also captured at this site (unpublished data), and the river itself is loaded with waste from upstream suburban areas with inadequate waste water systems (Badics et al. 2004), which may alter other environmental resource factors. Considering that most sites had large carabids present, but did not have a similar size class distribution, I would suggest that this is an important difference, and not an artifact of the sampling method which may retain large specimens because they have a more difficult time climbing the sides of the sampling cup to escape (Greenslade 1964).

Two of the species that were most numerous in the forest sites have traits that may provide interesting ecological insights – but their presence may also represent the negative impacts of exotic species. The exotic species found in the forests are voracious predators (Larochelle and Larivere 2003) and may out-compete or eliminate other beetles. In F2 and F4, only 3 and 6 species were found, respectively. *P. melanarius* was the most abundant species in F2, which is also the smallest of the park areas and therefore may be the least likely to maintain high species diversity in response to competitive pressure. A trait of *Pterostichus melanarius* is dimorphism: the proportion of flightless individuals increases as the species persists in a new habitat; it can be as low as 2% in stable habitats such as old forest patches, and 60-70% in newly established populations (Lovei and Sunderland 1996). This characteristic could prove useful to the understanding of the stability of the forest from the perspective of the animal inhabitants. However, *P. melanarius* is an exotic species that may disrupt the native carabid assemblages. The presence of the exotic *Carabus nemoralis* at F3, which contains 5 species, and F4 may also signal that this slug-loving large and effective colonist may be affecting the relative abundance of other species (Larochelle and Larivere 2003). Interestingly, *P. melanarius* and *C. nemoralis* accounted for over 80% (8,636 of 10,559) of the carabids captured in the urban forests of Edmonton, Canada (Niemelä et al. 2002). This suggests that these species, when present, may be having an impact on the structure of urban carabid communities.

The carabid assemblages of the garden plots generally include more exotic species, suggesting that the exotic plantings of the gardens also support exotic species assemblages, or that these species, much like exotic plant pioneer species, have become

established because of their ability to adapt in different or disturbed habitats. All species associated with gardens have the ability to fly, which suggests that adults may reach gardens through migration. Most beetles have eggs and larvae that develop in the soil, making them especially sensitive to soil disturbances caused by cultivation. In contrast, *Scarites subterraneus*, an abundant species in 3 of the 4 gardens lays its eggs on the leaves of trees or shrubs, encased in mud (Larochelle and Larivere 2003). This strategy may allow them to survive in cultivated areas, with high levels of ground disturbance.

The specific feeding habits of many carabids species is not well known, but the general feeding strategy of carabids has been shown to affect the pitfall capture rate (Greenslade 1964). Carnivorous carabids (which make up approximately 70% of the known carabid species) prefer open areas, such as those provided in the forests, assumedly because hunting is more efficient, while herbivorous species prefer greater plant diversity such as those provided in gardens (Harvey et al. 2008). The general feeding habits of the beetles (Larochelle and Larivere 2003) captured in these land uses appear to reflect this trend. Of the species with established eating habits, all those found in forests (*Agonum melanarium*, *Pterostichus mutus*, *Poecilus lucublandus* (*lucublandus*), *Pterostichus stygicus*, *Cyclotrachelus sodalis* (*sodalis*), *Chlaenius impunctifrons*, *Chlaenius tricolor*, *Pterostichus melanarius*, *Carabus nemoralis*, *Bembidion graciliforme*, *Pterostichus commutabilis*) are considered primarily carnivorous. The gardens hosted several species of carabids known to be at least occasionally herbivorous or seed predators (*Harpalus herbivagus*, *Amara aenea*, *Amara familiaris*, *Harpalus pennsylvanicus*, *Harpalus affinis*, *Scarites subterraneus*, *Diplocheila obtuse*, *Ophonus puncticeps*). At least four of the six species in the vacant lots are also omnivorous (*Scarites subterraneus*, *Diplocheila obtuse*, *Ophonus puncticeps*, *Badister parviceps*, *Anisodactylus rusticus*), and several are seed predators, which would make them well-adapted to the forage grass seed generally available in the vacant lots. The high activity-density of predatory beetles in forests may be due in part to their seeking of prey, which requires traveling greater distances and does not require herbaceous growth that could impede movement. The gardens also fostered relatively high activity-density, being amicable environments for herbivorous and omnivorous species. However, because the grasses prohibit movement, it is likely that few carabid species prefer the vacant lot

environment. Therefore it is not likely that the vacant lots have significantly more species present or exceed the species richness of the other habitats.

Patterns of species similarity showed that the forest habitats are significantly different from the gardens and vacant lots. The few expectations may be explained by the location or biology of several species. Some of the unexpected shared species in different habitat types may be a result of site proximity. For example, the sole carabid species that was found in both a vacant lot – V1 and in forest – F3 was *Cyclotrachelus sodalis* (*sodalis*). This species is common in disturbed, open habitats, is considered strongly favored by human activities, and is brachypterous - likely restricted to migrating on the ground (Larochelle and Larivere 2003). As most ground arthropod species are known to migrate only about one kilometer from their source population (Davis 1979; in McIntyre 2000), it is possible that the source of *C. sodalis* is F3 [River Rouge Park], which at its northern boundary is less than a kilometer to V1. Another singular shared species is that of *Chlaenius tricolor*, which was found in G2 [Earthworks] and F4 [Palmer Park]. *Chlaenius tricolor* is generally found near water bodies, such as the pond in F4, and in cultivated areas such as the G2 production area; it is also favored by human activities (Larochelle and Larivere 2003). As a much greater distance than 1 km separates these two sites, it is more likely that this species is more gregarious in the city.

Habitat Characteristic Association

Descriptive characteristics for study sites are often assumed when the “dummy” variables used to define the urban-rural gradient (i.e. rural, suburban, urban). Defining the internal and external site environmental characteristics is more beneficial in determining valid correlations and allowing for experimental replication (McIntyre 2000; Deichsel 2006; Smith et al. 2006). Such measures can also be challenge to analyze because of the complexity of the environment and interdependence of many physical factors.

Using the backwards regression model did not yield many relationships between site size, structure or composition that seem to be predictive of the activity-density or species richness; this is not unique among urban carabid studies (Small, Sadler, and Telfer 2006; de Groot, Kleijn, and Jogan 2007; e.g. Eyre 2004; Gaublomme et al. 2008). The significance of vegetative groundcover at the one hectare scale suggest that the

matrix around the site is of import; other recent research supports that the degree of isolation is a primary characteristic that is related to biodiversity (Small, Sadler, and Telfer 2006; Deichsel 2006). As the 1 m² scale percent cover variables are all dependent (i.e., the percent of cover by rocks will likely affect the percent cover by grass); this is likely an example of spurious correlations due to the relative difficulty of separating environmental variables that are dependent on one another. In general, the significant predictors in these regressions are representative of the great difference in the forests and other habitats. The forests are covered by tree canopy, and therefore have a high percentage of shade, while the vacant lots and gardens are primarily unshaded. The vacant lots, which had a much smaller activity-density than the other sites, are maintained by mowing, encouraging the European grazing grasses that have persisted under this management (unpublished data, this study).

A closer inspection of relationships that are suggested in the morpho-ecological analysis or specific species biological traits, such as the relationship between site age and the presence of non-flying species or the relationship of shrub and tree cover type to *Scarites subterraneus* presence and activity-density may yield results that are more constructive towards developing the indicator utility of carabid beetles in this urban environment.

Historical comparisons

Unfortunately, estimates of biodiversity based on museum collection specimens has been shown to drastically underestimate the actual species richness (Petersen, Meier, and Larsen 2003), so no such effort was made with this data. Though there are no direct implications for the historical information at this time, the field notebooks of the collectors and other resources (Hubbard, Schwarz, and LeConte 1878) may serve to provide broader perspective of the changes in the carabid communities over time. As several of the species represented in the garden samples were also found in the historical collection, it suggests that the forest areas may have once included these species as the early collectors tended to gather specimens in natural areas.

Implications for Management and Research

It would be beneficial to have a greater understanding of the degree of isolation of habitat types, especially the forests that appear to have conservation value. Significant work with GIS mapping has been conducted by the City of Detroit Planning Department, American Forests (2006), and at University of Michigan that could assist in spatial analysis (Zhao, Brown, and Bergen 2007) that could establish a more concise and accurate depiction of the urban matrix than the current data set affords. In addition to contributing to future ecological studies, such information could assist city planners in linking forest sites through greenway development and land managers in conservation considerations.

In Detroit, associating carabid morphoecological and biodiversity data with existing forest data, or ground or water pollution, may help to identify species that could be bioindicators. For instance, *C. sodalis* may be a candidate for studies of carabid beetle migration in the northwestern area of Detroit, and *P. melanarius* may provide interesting insight into establishment of exotic species. Though it was not the intent of this study to advocate for carabid beetle conservation in particular, if carabids are found to be representative of general biodiversity in areas, the implications for their conservation when confronting climate change or other environmental issues may likewise benefit other species. Urban areas are potentially representative of the effects of future climate changes, as impermeable ground cover causes “heat islands” with warmer temperatures compared to surrounding non-urban habitat (Wilby and Perry 2006; Brazel et al. 2000). An analysis of the carabid response to warming during the Quaternary Period showed relatively little loss of beetle species – but the winged migration that allowed beetles to survive these past changes will be increasingly difficult with increasing habitat fragmentation, and we are likely to witness extinctions beyond that caused by anthropogenic habitat alteration or warming alone (Ashworth 2001).

Alternative sampling methods such as hand collection, quadrat sampling, and larvae identification may provide for a more complete biological picture that confirms the trends suggested in this survey (Petrillo 2006; Greenslade 1964). A “nested cross array” of trapping, which increases the space between traps that are placed on two perpendicular transects from the center to the outer traps, has been found to be a better pitfall sampling scheme for estimating a more accurate density of carabids at a site; however, this

sampling method requires a nearly homogeneous portion of the habitat and that the outer traps away from habitat edges (Perner and Schueler 2004). This would be difficult in forests that have relatively small areas and irregular topography such as vernal pools in the Detroit forests. However, sampling across a greater variety of these forest microhabitats which may have been missed and considering other factors, such as downed wood that are known to be preferred habitats for some would also be important if seeking a more complete knowledge of the carabid assemblage carabids (Petrillo 2006; Deichsel 2006; Lovei and Sunderland 1996; Work et al. 2002).

The carabid data also suggests that multiple measures are valuable in management decisions. Despite having the lowest FQI, Belle Isle proved to be the most species rich and free of exotic carabid species. Further research describing the biota of Detroit may be fundamental to long-term or broad-scale conservation efforts of the unique forests in the city. In order to foster biological communities, experimental ecological studies that examine the disturbance regimes and resources that affect the selected communities would also be indispensable (Shochat et al. 2006)

Conclusion

The findings of this study indicate that the remnant forests do indeed represent important vestiges for biological diversity. The carabid communities of remnant urban forests are distinct from those of either the vacant lots or gardens. Forests are the home to more species and individuals of carabid beetles than either of the more urban greenspace uses. Morphological characteristics such as flightless species and large-bodied species are also more prominent in the forests, suggesting less environmental disturbance and greater resource availability. The gardens and vacant lot carabid communities were more similar to each other in species composition; however, many more individuals were found in the gardens than in the vacant lots, which may indicate that gardens have resources that are preferred by carabids in comparison with vacant lots.

Most cities throughout the world are growing, others, like Detroit, are in a postindustrial phase characterized by a population decline. Furthermore, there is significant discussion about the need to develop sustainable cities with more green spaces and local food production. As the urban areas keep evolving, it is important that we

understand the long-term effects of humans in the environment, and apply this knowledge to the planning and management of urban areas. In order to do this, economical and efficient sampling methods need to be developed. Though the general measures of species richness and abundance may not be appropriate to identify the type of habitat, or variation among habitat types, the morphological characteristics of the carabid assemblage seem to provide information that may give insights to patterns of distribution and processes, such as species migration and environmental similarities that are not represented in other current management tools.

Further investigation of carabid assemblages could shed light on additional species or relationships that may be valuable indicators of other environmental elements, such as climate changes, or simply assist the human population with a better understanding of the biodiversity changes in urban environments that have undergone a significant rise and fall in anthropogenic activity.

A clearly defined management goal, coupled with an understanding of the biological characteristics of the carabid species is important to the utility of research efforts in forests, gardens and vacant areas of the Detroit. While conservation is a goal in parks such as Belle Isle, improved biocontrol by encouraging carabid presence may be an interest of gardeners, while fostering more native herbaceous plants and tree cover in vacant lots might benefit both the local socioeconomic and biological components of the urban setting. Continued research may bring insights that allow for the implementation of low-cost, beneficial ecological management techniques that are needed to restore and improve urban areas.

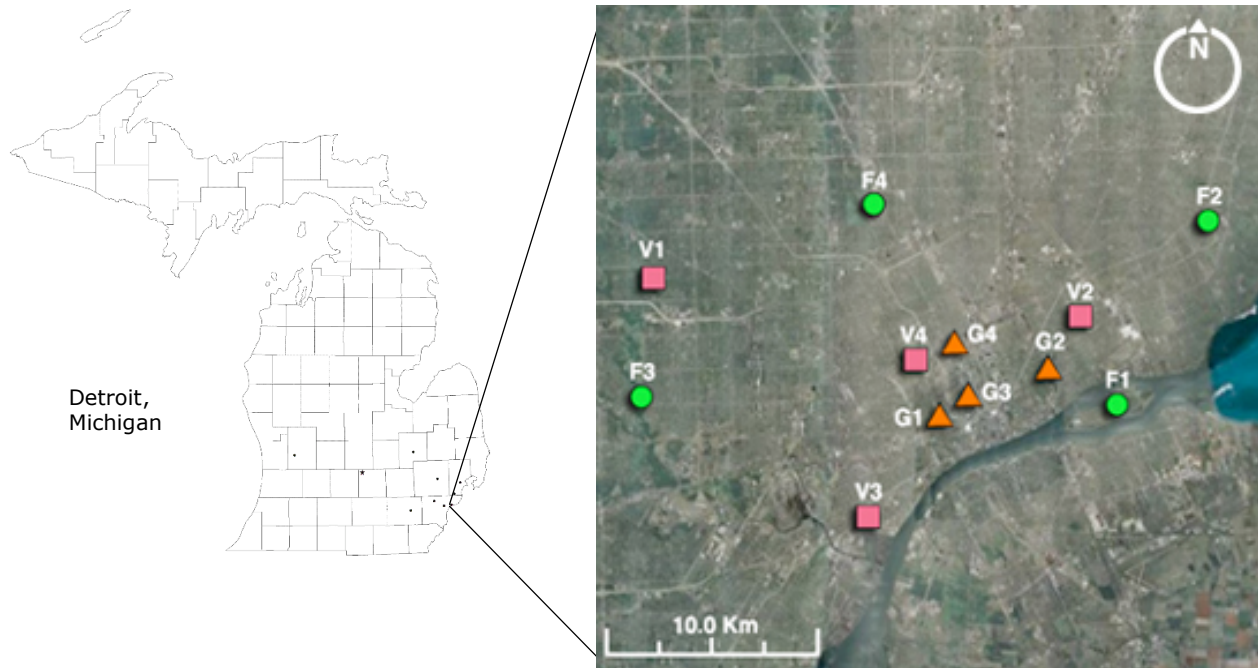


Figure 1. Location of Detroit, Michigan and the 12 site study sites

Table 1. Site Descriptions

| Site Name | Site Location | Area (m ²) | Lat/Long coordinates | |
|-------------------|--|------------------------|----------------------|---------------|
| Garden 1 (G1) | Hope Takes Root, Wabash St at Perry St | 1830 | 42°20'9.14"N | 83° 4'40.78"W |
| Garden 2 (G2) | Earthworks, near 1264 Meldrum St | 2665 | 42°21'15.69"N | 83° 0'44.11"W |
| Garden 3 (G3) | Birdtown Garden, near 3443 Cass Ave | 828 | 42°20'40.50"N | 83° 3'36.72"W |
| Garden 4 (G4) | Acres of Hope, 5930 Woodward Ave | 353 | 42°21'50.51"N | 83° 4'8.64"W |
| Vacant lot 1 (V1) | 20336-20330 Lyndon St | 706 | 42°20'29.48"N | 82°58'28.92"W |
| Vacant lot 2 (V2) | 4174-4180 McClellan St | 306 | 42°25'7.16"N | 82°55'18.93"W |
| Vacant lot 3 (V3) | 8366 Thaddeus St | 306 | 42°20'51.30"N | 83°14'51.10"W |
| Vacant lot 4 (V4) | 5773 Wabash St | 520 | 42°25'27.66"N | 83° 6'55.30"W |
| Forest 1 (F1) | Belle Isle Park | 737500 | 42°23'36.78"N | 83°14'27.81"W |
| Forest 2 (F2) | Balduck Park | 28500 | 42°22'31.42"N | 82°59'45.74"W |
| Forest 3 (F3) | River Rouge Park | 368500 | 42°17'32.52"N | 83° 7'7.79"W |
| Forest 4 (F4) | Palmer Park | 258000 | 42°21'25.01"N | 83° 5'23.12"W |

Table 2. Average Values for Habitat Characteristics

| | Gardens | Vacant Lots | Forests |
|---|---------|-------------|---------|
| 1,000 m² (1 ha) | | | |
| % impermeable (concrete/asphalt) | 22.00 | 22.50 | 1.25 |
| % vegetation | 39.00 | 70.00 | 21.25 |
| % building area | 20.75 | 7.75 | 0 |
| % shrubs | 4.75 | 1.00 | 32.50 |
| % bare ground | 3.00 | 1.25 | 3.00 |
| # trees [greater than 30cm in circumference] | 30.75 | 45.50 | 321.75 |
| # buildings | 3.75 | 12.75 | 0 |
| 400 m² | | | |
| Average % Canopy Cover | 5.41 | 10.79 | 92.54 |
| Richness of woody species | 1.25 | 2.50 | 11.75 |
| Tree density (trees/m ²) | 0.00 | 0.01 | 0.03 |
| Tree height (m) | 0.41 | 9.09 | 14.60 |
| Tree circumference | 7.75 | 106.52 | 111.21 |
| Shrub [1-3m tall] density (shrubs/m ²) | 0.02 | 0.00 | 0.24 |
| Shrub height | 0.90 | 0.73 | 1.42 |
| Shrub circumference | 1.30 | 1.16 | 1.73 |
| 1 m² | | | |
| Average of % cover grasses | 21.19 | 81.60 | 1.88 |
| Average of % cover bare ground | 30.22 | 6.44 | 1.60 |
| Average of % cover forbs/herbs | 38.10 | 10.19 | 21.35 |
| Average of % cover rocks/wood panels | 1.38 | 1.63 | 6.25 |
| Average of % cover by leaf litter | 0.00 | 74.53 | 86.81 |
| Average of % cover by mulch | 8.88 | 0.00 | 0.00 |
| Average of % cover by fallen branches | 1.44 | 1.19 | 8.32 |
| Average height of tallest non-woody vegetation (cm) | 29.06 | 38.00 | 28.63 |

Table 3. Temperature (°C) and Precipitation (cm) during sampling days
(The Weather Exchange 2007)

| Date (2007) | Mean Temperature | High Temperature | Low Temperature | Precipitation |
|------------------------|-----------------------------|-----------------------------|----------------------------|----------------------|
| May 15 | 22 | 30 | 12 | 1.07 |
| May 16 | 11 | 13 | 7 | 0.18 |
| May 17 | 10 | 12 | 5 | 0.13 |
| May 18 | 11 | 13 | 7 | 0.18 |
| June 16 | 22 | 28 | 15 | 0 |
| June 17 | 25 | 30 | 21 | 0 |
| June 18 | 26 | 33 | 21 | 0 |
| June 19 | 24 | 27 | 18 | 0.10 |
| July 16 | 22 | 27 | 16 | 0 |
| July 17 | 24 | 22 | 18 | 0.18 |
| July 18 | 24 | 30 | 18 | 0.03 |
| July 19 | 22 | 28 | 17 | 0.81 |
| August 15 | 23 | 26 | 21 | 0 |
| August 16 | 26 | 30 | 22 | 0 |
| August 17 | 22 | 27 | 17 | 0 |
| August 18 | 16 | 21 | 12 | 0.03 |

Table 4. Forest Characteristics

| Forest | Forest Type c. 1800 (Comer et al. 1995) | Floristic Quality Index ** (FQI) | Contemporary Community Type |
|----------------------------|--|---|---|
| Balduck Park (F2) | Mixed Hardwood Swamp | 23.36* | Wetland Oak Forest* |
| Belle Isle (F1) | Beech-Sugar Maple Forest | 27.57* | Deciduous Swamp Community [†] / Wet- mesic Flatwood [Ⓞ] |
| Palmer Park (F4) | Mixed Hardwood Swamp | 39.98* | Combination of: Southern Swamp, Mesic Southern Forest, Dry-mesic Southern Forest* |
| River Rouge Park F3) | Beech-Sugar Maple Forest | 49.50 [†] | N/A [old-growth floodplain forest] |

Compiled from *(Weatherbee and Klatt 2004), (King & MacGregor Environmental 2002), and
[†](King & MacGregor Environmental 2004) with community descriptions determined from
[†](Barnes 2004) and [Ⓞ](Kost 2007)

**FQI = average conservation coefficient of plant species, divided by the square root of the total number of species found in botanical surveys. A FQI > 35 possess sufficient conservatism and richness that they are floristically important from a statewide perspective. Areas > 50 and higher are extremely rare and are considered “to represent a significant component of Michigan’s native biodiversity and natural landscapes” (Herman et al. 2001).

Table 5. Total individuals and species of carabid beetles and the percent of carabid species found in morpho-ecological and historical categories by habitat and site

| | Vacant | | | G1 | G2 | G3 | G4 | F1 | F2 | F3 | F4 | V1 | V2 | V3 | V4 |
|--|---------|-------|---------|-------|--------|-------|--------|--------|-------|-------|-------|--------|--------|--------|--------|
| | Gardens | Lots | Forests | | | | | | | | | | | | |
| Individuals | 79 | 13 | 105 | 6 | 32 | 16 | 25 | 18 | 24 | 41 | 22 | 1 | 1 | 5 | 6 |
| Species Richness | 11 | 6 | 18 | 2 | 5 | 6 | 3 | 11 | 3 | 5 | 6 | 1 | 1 | 4 | 3 |
| Length | | | | | | | | | | | | | | | |
| Species < 7mm | 27.27 | 16.67 | 22.22 | 0.00 | 40.00 | 33.33 | 0.00 | 36.36 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 25.00 | 0.00 |
| Species ≥ 7mm < 15mm | 54.55 | 50.00 | 50.00 | 50.00 | 40.00 | 33.33 | 66.67 | 54.55 | 66.67 | 20.00 | 66.67 | 0.00 | 100.00 | 50.00 | 66.67 |
| Species ≥ 15mm | 18.18 | 33.33 | 27.78 | 50.00 | 20.00 | 33.33 | 33.33 | 9.09 | 33.33 | 80.00 | 33.33 | 100.00 | 0.00 | 25.00 | 33.33 |
| Flight Ability | | | | | | | | | | | | | | | |
| Brachypterous | 0.00 | 16.67 | 16.67 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 60.00 | 33.33 | 100.00 | 0.00 | 0.00 | 0.00 |
| Dimorphic | 9.09 | 0.00 | 11.11 | 0.00 | 0.00 | 16.67 | 0.00 | 9.09 | 33.33 | 20.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Macropterous/ Submacropterous | 90.91 | 83.33 | 72.22 | 100. | 100.00 | 83.33 | 100.00 | 90.91 | 66.67 | 20.00 | 66.67 | 0.00 | 100.00 | 100.00 | 100.00 |
| North American Origin | 63.64 | 83.33 | 88.89 | 50.00 | 60.00 | 83.33 | 66.67 | 100.00 | 66.67 | 60.00 | 83.33 | 100.00 | 100.00 | 75.00 | 66.67 |
| Represented in Historical Collection | 63.64 | 33.33 | 33.33 | 50.00 | 60.00 | 50.00 | 100.00 | 18.18 | 33.33 | 40.00 | 66.67 | 0.00 | 0.00 | 25.00 | 33.33 |

Origin: 0=North America or United States, 1=Europe (Bousquet and Laroche 1993; Thiele 1977; Lovei and Sunderland 1996; Laroche and Larivere 2003)

Length: from literature (Laroche and Larivere 2003)

Dispersal (flight ability): Macropterous (frequent flyer); Submacropterous (occasional flyer); Dimorphism (flight ability varies with stability of environment); Brachypterous (Does not fly) (Laroche and Larivere 2003)

Historical (appearance in collection at U-M; N=no specimens from Detroit or Wayne county; date=first captured specimen of collection)

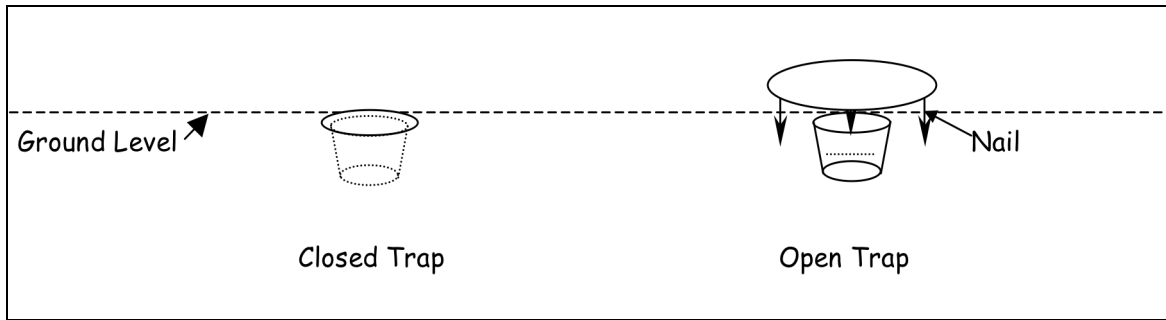


Figure 2. Pitfall Trap Design. Closed traps (left) consist of a 500ml clear plastic cup, established just below ground level and covered when not sampling; open traps (right) have another plastic cup nested in the top cup, just at ground level, and were covered with a plastic plate in order to keep out debris and precipitation.

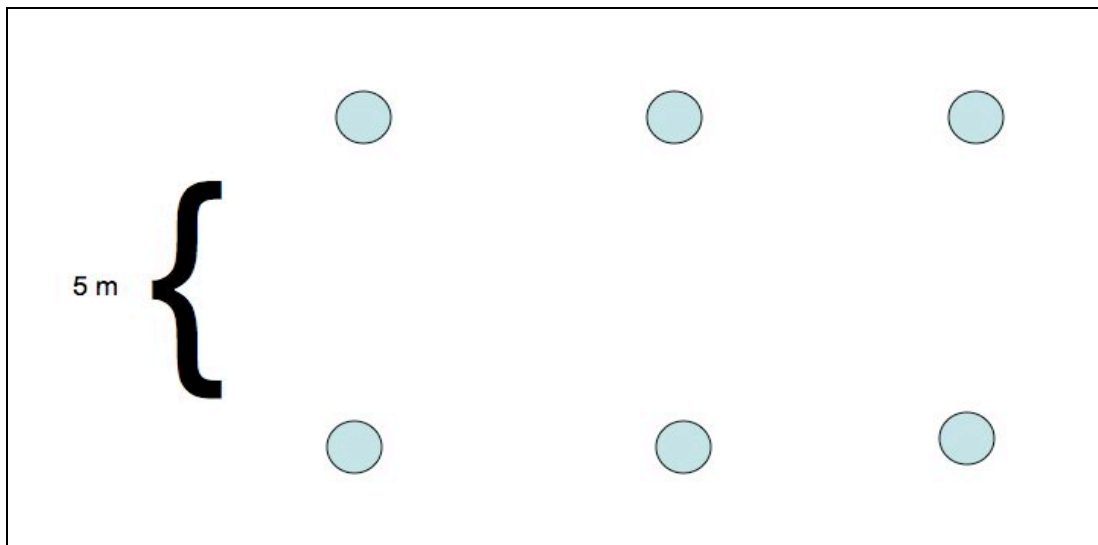


Figure 3. Diagram of pitfall station layout, showing two transects of three pitfall stations, separated by 5m each, with the two transects separated by 5m to form a rectangle.

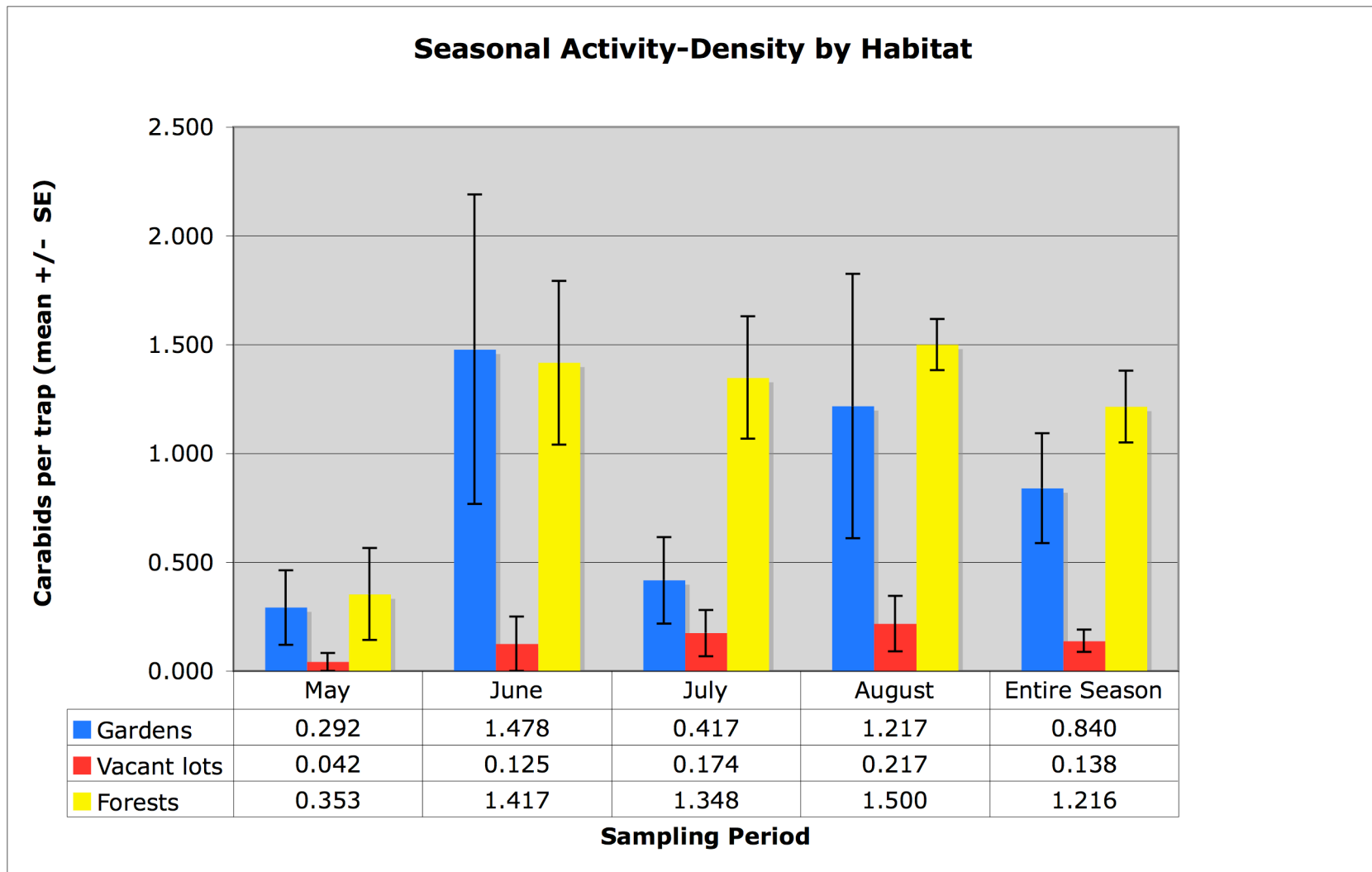


Figure 4. Seasonal activity-density of carabids in each habitat for the four sampling months and over the entire season

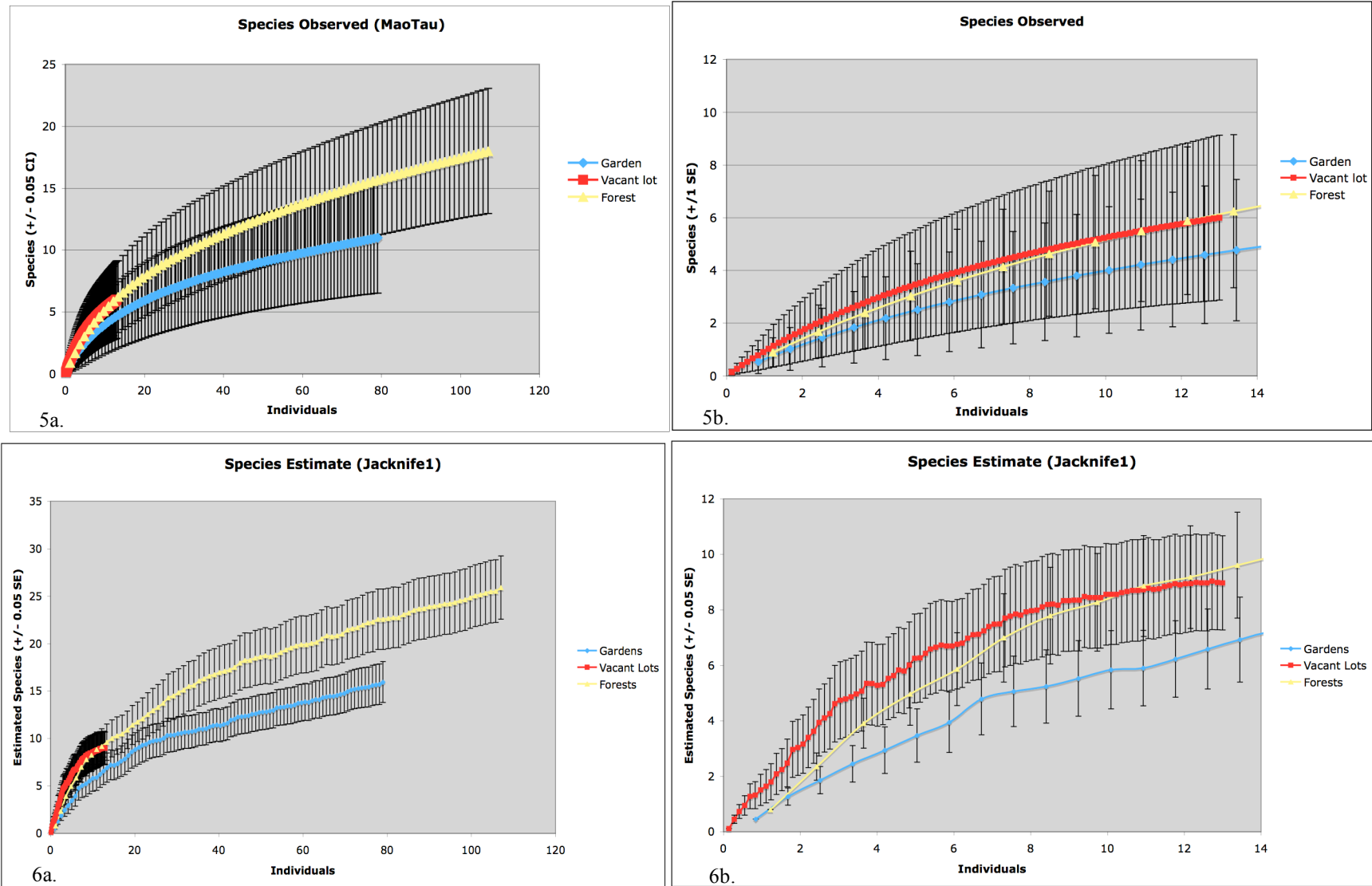


Figure 5a. The observed species rarefaction curve of carabids in each habitat; Figure 5b. the portion of the rarefaction curve where all habitats are represented; Figure 6a. The species estimation for each habitat; Figure 6b. the portion of the species estimate curve where all habitats are represented.

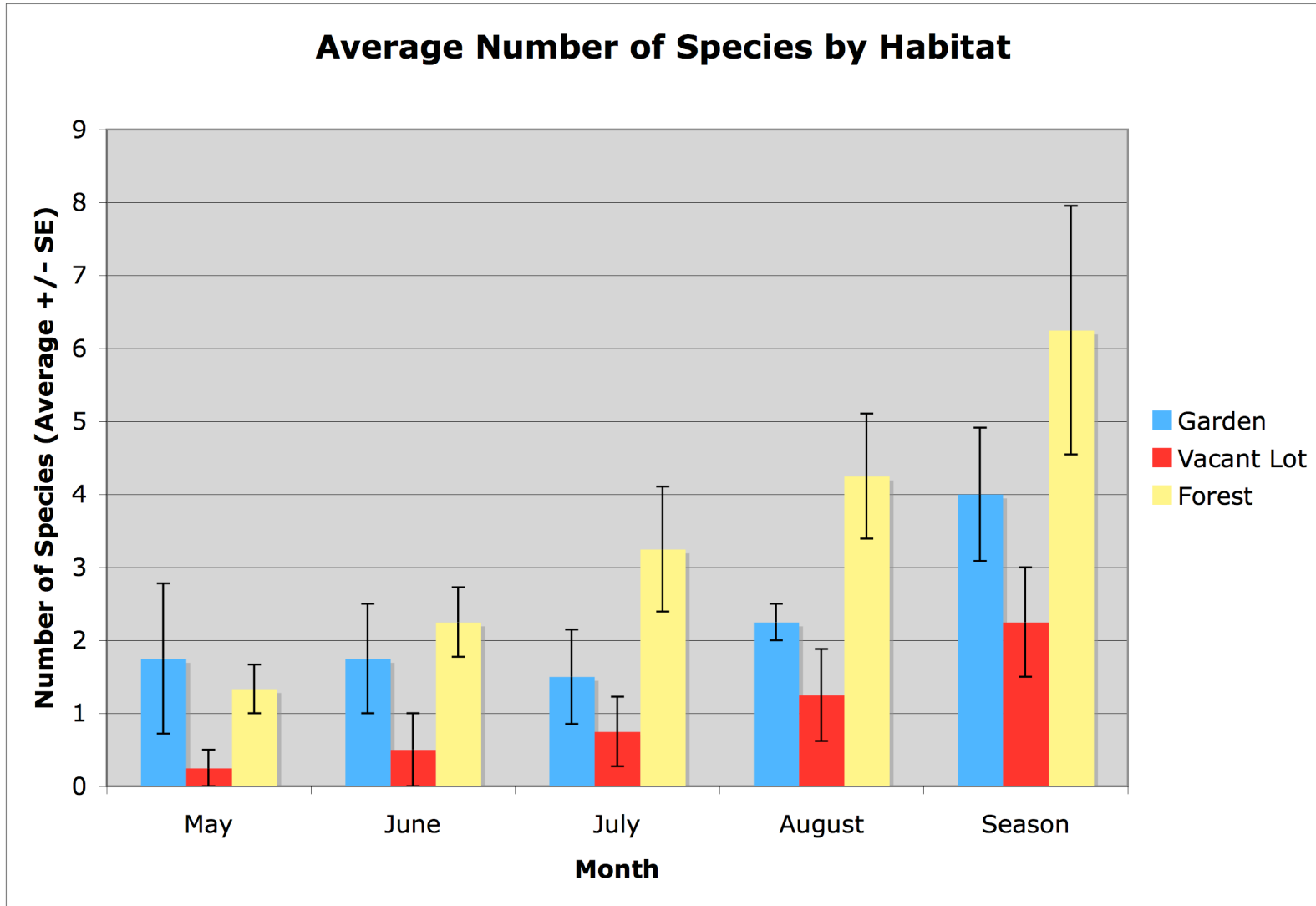


Figure 7. Average number of species captured in each habitat for month and the total for the season

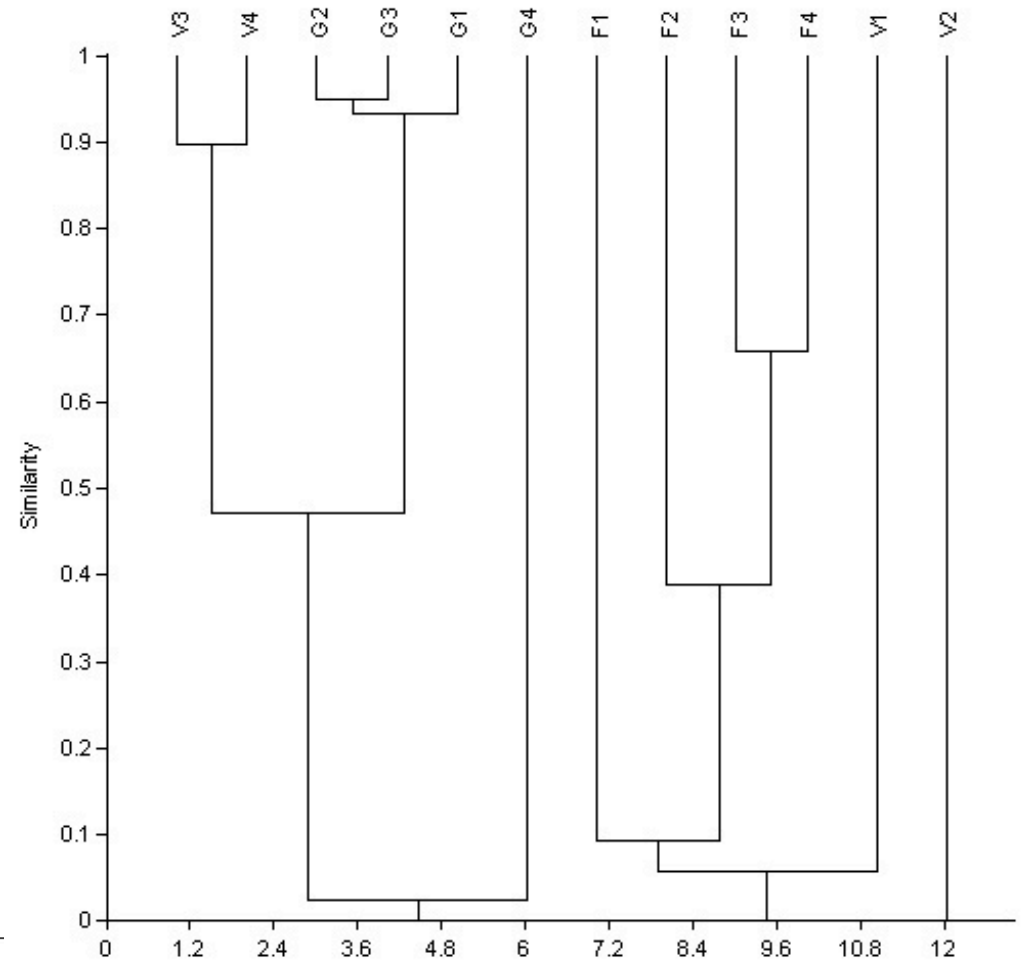
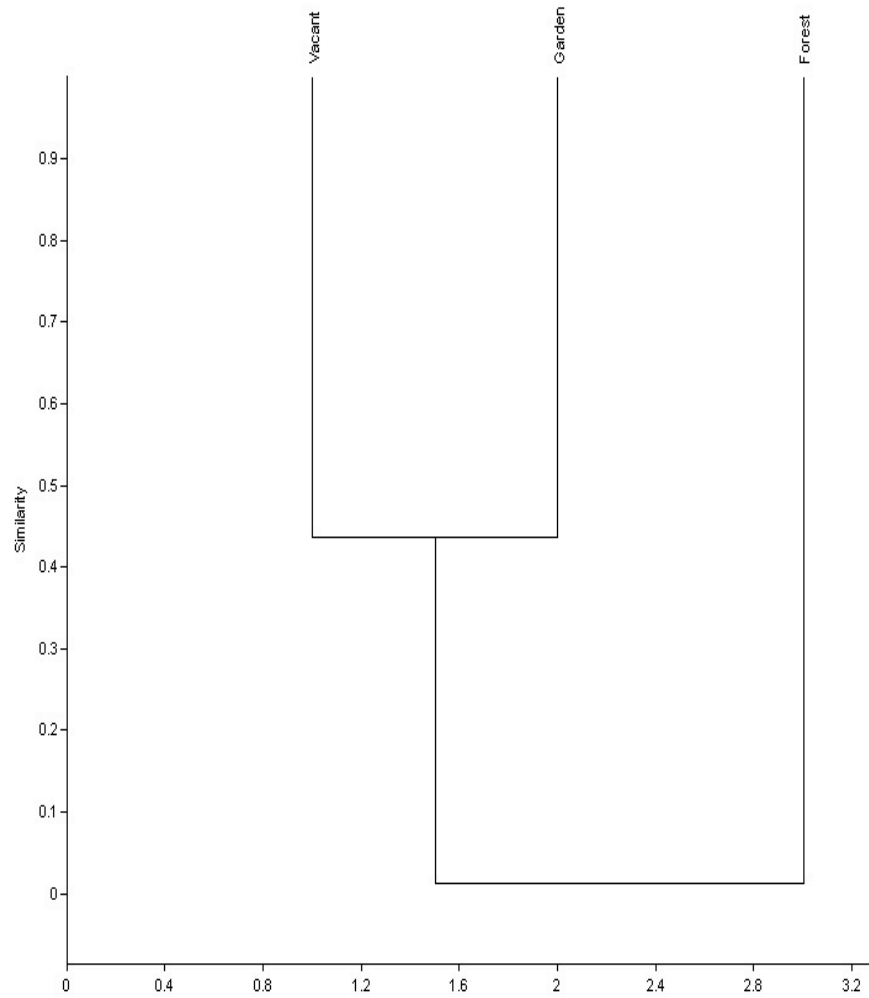


Figure 8. Cluster analysis of carabid species abundance for habitat types.

Figure 9. Cluster analysis of carabid species abundance for sites.

Results are based on the Morista Similarity Index and the UPGMA agglomerative method.

Table 6. Morista Similarity Index Values

| Pairwise Comparison | Morista Index Value |
|-------------------------|---------------------|
| Gardens | |
| G1 vs. G2 | 0.918 |
| G1 vs. G3 | 0.944 |
| G1 vs. G4 | 0 |
| G2 vs. G3 | 0.947 |
| G2 vs. G4 | 0 |
| G3 vs. G4 | 0.122 |
| Vacant lots | |
| V1 vs. V2 | 0 |
| V1 vs. V3 | 0 |
| V1 vs. V4 | 0 |
| V2 vs. V3 | 0 |
| V2 vs. V4 | 0 |
| V3 vs. V4 | 0.897 |
| Forests | |
| F1 vs. F2 | 0.117 |
| F1 vs. F3 | 0.039 |
| F1 vs. F4 | 0.119 |
| F2 vs. F3 | 0.245 |
| F2 vs. F4 | 0.529 |
| F3 vs. F4 | 0.658 |
| Gardens vs. Vacant Lots | 0.436 |
| Gardens vs. Forests | 0.001 |
| Vacant Lots vs. Forests | 0.022 |

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| Appendix A. Site Habitat Characteristics | | | | | | | | | | | | |
|--|--------|--------|--------|--------|---------|--------|--------|---------|---------|---------|---------|---------|
| Site | G1 | G2 | G3 | G4 | V1 | V2 | V3 | V4 | F1 | F2 | F3 | F4 |
| 1 ha scale | | | | | | | | | | | | |
| % impermeable (concrete/asphalt) | 15.000 | 25.000 | 33.000 | 15.000 | 25.000 | 30.000 | 15.000 | 20.000 | 0.000 | 0.000 | 0.000 | 5.000 |
| % vegetation | 70.000 | 55.000 | 28.000 | 3.000 | 65.000 | 60.000 | 75.000 | 80.000 | 10.000 | 25.000 | 25.000 | 25.000 |
| % building area | 10.000 | 15.000 | 33.000 | 25.000 | 10.000 | 10.000 | 10.000 | 1.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| % shrubs | 10.000 | 5.000 | 1.000 | 3.000 | 1.000 | 1.000 | 1.000 | 1.000 | 30.000 | 25.000 | 25.000 | 50.000 |
| % bare ground | 1.000 | 1.000 | 5.000 | 5.000 | 1.000 | 1.000 | 2.000 | 1.000 | 5.000 | 1.000 | 1.000 | 5.000 |
| # trees greater than 30cm in circumference | 34.000 | 38.000 | 13.000 | 38.000 | 63.000 | 41.000 | 53.000 | 25.000 | 179.000 | 412.000 | 366.000 | 330.000 |
| # buildings | 2.000 | 4.000 | 4.000 | 5.000 | 16.000 | 15.000 | 16.000 | 4.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 400 m2 | | | | | | | | | | | | |
| Avg Canopy Cover (%) | 7.396 | 0.708 | 12.760 | 0.781 | 11.667 | 7.865 | 17.292 | 6.319 | 91.458 | 91.333 | 94.236 | 93.125 |
| richness of woody species | 1.000 | 0.000 | 1.000 | 3.000 | 0.000 | 7.000 | 1.000 | 2.000 | 8.000 | 17.000 | 7.000 | 15.000 |
| tree density (3trees/400m2) | 0.0000 | 0.0000 | 0.0000 | 0.0050 | 0.0000 | 0.0200 | 0.0100 | 0.0025 | 0.0175 | 0.0450 | 0.0200 | 0.0250 |
| Avg tree height | 0.000 | 0.000 | 0.000 | 1.650 | 0.000 | 9.750 | 6.625 | 20.000 | 9.500 | 12.526 | 18.778 | 17.600 |
| Avg tree circumference | 0.000 | 0.000 | 0.000 | 31.000 | 0.000 | 66.813 | 40.250 | 319.000 | 104.857 | 74.158 | 136.333 | 129.500 |
| shrub density | 0.0175 | 0.0000 | 0.0050 | 0.0375 | 0.0000 | 0.0075 | 0.0000 | 0.0025 | 0.1875 | 0.4425 | 0.0375 | 0.3025 |
| Avg shrub height | 1.051 | 0.000 | 1.110 | 1.447 | 0.000 | 1.600 | 0.000 | 1.300 | 1.460 | 1.480 | 1.410 | 1.349 |
| Avg shrub circumference | 1.186 | 0.000 | 0.850 | 3.177 | 0.000 | 1.833 | 0.000 | 2.800 | 1.954 | 1.597 | 1.913 | 1.470 |
| 1 m2 | | | | | | | | | | | | |
| Average of % cover grasses | 40.130 | 7.880 | 0.500 | 36.250 | 80.000 | 68.880 | 86.250 | 91.250 | 6.250 | 1.250 | 0.000 | 0.000 |
| Average of % cover bare ground | 21.750 | 22.500 | 68.750 | 7.880 | 0.000 | 22.500 | 1.380 | 1.880 | 2.500 | 2.630 | 1.250 | 0.000 |
| Average of % cover forbs/herbs | 24.630 | 69.250 | 4.750 | 53.750 | 22.750 | 5.130 | 7.750 | 5.130 | 0.130 | 1.500 | 13.000 | 70.750 |
| Average of % cover rocks/wood panels | 3.880 | 0.000 | 1.250 | 0.380 | 0.000 | 5.000 | 0.250 | 1.250 | 0.000 | 25.000 | 0.000 | 0.000 |
| Average of % cover by leaf litter | 0.000 | 0.000 | 0.000 | 0.000 | 100.000 | 25.130 | 75.000 | 98.000 | 88.750 | 72.250 | 98.750 | 87.500 |
| Average of % cover by mulch | 11.500 | 0.250 | 16.250 | 7.500 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Average of % cover by fallen branches | 5.750 | 0.000 | 0.000 | 0.000 | 1.630 | 2.000 | 0.000 | 1.130 | 12.500 | 7.750 | 7.880 | 5.130 |
| Average of height of tallest non-woody vegetation (cm) | 46.250 | 27.750 | 30.480 | 11.750 | 54.750 | 21.500 | 31.000 | 44.750 | 34.750 | 31.750 | 18.750 | 29.250 |

Appendix B. Carabid Species

| Species | Origin | Length (mm) | Dispersal | Earliest Historical Specimen | Total Individuals for Habitat Type | | |
|---|--------|----------------|-----------|---------------------------------|------------------------------------|------------|--------|
| | | | | | Garden | Vacant Lot | Forest |
| <i>Agonum ferreum</i> Haldeman | N | 8 | M | N | 0 | 0 | 3 |
| <i>Agonum melanarium</i> Dejean | N | 10 | M | 1907 | 0 | 0 | 5 |
| <i>Amara aenea</i> DeGeer | EU | 8.5 | M | N | 3 | 0 | 0 |
| <i>Amara familiaris</i> Duftschmidti | EU | 6.5 | M | N | 5 | 0 | 0 |
| <i>Anisodactylus rusticus</i> Say | N | 12 | M | 1913 | 0 | 1 | 0 |
| <i>Badister parviceps</i> Ball | N | 5 | M | N | 0 | 1 | 0 |
| <i>Bembidion affine</i> Say | N | 4 | M | N | 0 | 0 | 1 |
| <i>Bembidion frontale</i> LeConte | N | 2.5 | M | N | 0 | 0 | 1 |
| <i>Bembidion graciliforme</i> Hayward | N | 5.5 | M | N | 0 | 0 | 3 |
| <i>Bradycellus kirbyi</i> Horn | N | 7 | M | N | 0 | 0 | 1 |
| <i>Carabus nemoralis</i> Mueller | EU | 22 | B | 1938 | 0 | 0 | 14 |
| <i>Chlaenius impunctifrons</i> Say | N | 15 | M | N | 0 | 0 | 1 |
| <i>Chlaenius tricolor</i> Dejean | N | 12 | M | 1907 | 1 | 0 | 2 |
| <i>Cyclotrachelus sodalis</i> (sodalis) LeConte | N | 17 | B | N | 0 | 1 | 6 |
| <i>Diplocheila obtusa</i> LeConte | N | 12 | M | N | 1 | 5 | 0 |
| <i>Elaphropus anceps</i> LeConte | N | 3 | M | 1910 | 3 | 0 | 0 |
| <i>Harpalus affinis</i> Schrank | EU | 10.5 | M | 1919 | 10 | 0 | 0 |
| <i>Harpalus herbivagus</i> Say | N | 9 | M | 1920 | 1 | 0 | 0 |
| <i>Harpalus pensylvanicus</i> DeGeer | N | 15 | M | 1919 | 16 | 0 | 0 |
| <i>Ophonus puncticeps</i> Stephens | EU | 9 | M | N | 1 | 2 | 0 |
| <i>Patrobus longicornis</i> Say | N | 12 | D | 1906 | 0 | 0 | 1 |
| <i>Poecilus lucublandus</i> (lucublandus) Say | N | 13.5 | SM | N | 0 | 0 | 28 |
| <i>Pterostichus commutabilis</i> Motschulsky | N | 9 | M | N | 0 | 0 | 1 |
| <i>Pterostichus melanarius</i> Illiger | EU | 19 | D | N | 0 | 0 | 14 |
| <i>Pterostichus mutus</i> Say | N | 13 | M | 1911 | 0 | 0 | 1 |
| <i>Pterostichus praetermissus</i> Chaudoir | N | 9 | M | N | 0 | 0 | 1 |
| <i>Pterostichus stygicus</i> Say | N | 16.5 | B | 1938 | 0 | 0 | 22 |
| <i>Scarites subterraneus</i> Fabricius | N | 18 | M | cir. 1900 | 37 | 3 | 0 |
| <i>Stenolophus conjunctus</i> Say | N | 4 | D | 1909 | 1 | 0 | 0 |
| <i>Trichotichnus autumnalis</i> Say | N | 9 | M | N | 0 | 0 | 2 |

CHAPTER III

BEETLE (COLEOPTERA) ASSEMBLAGES IN GREENSPACES OF DETROIT

Despite the increasing land area devoted to human habitation, there is a great lack of information available for ecologically based management (Connor et al. 2003). Urban areas consist of dynamic areas of land that are heterogeneous in nature (Shochat et al. 2006), and are therefore a prime setting for considering the migration and persistence of species. The number and biomass of insects and other invertebrates far outnumber that of any vertebrate taxa; yet despite their presence in almost every urban environment, invertebrates such as arthropods tends to be the least well-studied of the fauna (Luniak 2008). In addition to being abundant, invertebrates are crucial to environmental functions of the urban environment, such as nutrient cycling, consumption of plant matter and detritus, consumption of other invertebrate “pest” species, pollination and serving as the prey of many other animals (Luniak 2008).

McIntyre calls attention to the gaps in research of the ecology of urban arthropods (2000), and provides convincing arguments for studying arthropods in urban settings: 1) as a diverse taxon, they may provide a “snapshot” of overall biological diversity; 2) short generation times quickly reflect anthropogenic changes in soil and vegetation; 3) they are fairly easy and socially acceptable to sample; 4) they represent a range of trophic levels; and 5) they may be important sociological, agronomical, and economical components of human-altered habitats.

The beetles (Coleoptera) are the most numerous order of animals on the earth, comprising approximately a quarter of all named animal species, and are of particular interest as they represent a range of functions in the environment, from second-order consumers such as ground beetles to mycophages such as minute fungus beetles (Marshall 2006). Approximately one-quarter of urban arthropod studies have been devoted to beetles (McIntyre 2000).

Carabid beetles (Coleoptera: Carabidae) have become the standard taxonomic candidate for researchers examining the impact of urban areas on surrounding forests or natural areas (Niemelä and Kotze 2000; Niemelä et al. 2002). However, the biological

information revealed by carabids is limited (Deichsel 2006). Key biodiversity indicators such as carabid species richness do not appear to be correlated with the degree of urbanization (Niemelä et al. 2002). Though a sustained richness or density along a rural-urban gradient provides for study of other characteristics of assemblages, significant relationships between habitat characteristics and carabid assemblages have not been identified (Small, Sadler, and Telfer 2006). However, in a study of carabid and rove (Coleoptera: Staphylinidae) beetles, the rove beetles showed increasingly significant differences in species assemblages as the matrix grew increasingly urban, while carabid beetles did not (Deichsel 2006). Climate variables also have a significant effect on carabid capture rate, and due to the relatively small number of species per assemblage, climactic effects can have a strong impact on the results of carabid studies (Deichsel 2006). These factors suggest that it may be necessary to take a broader set of organisms into consideration to have a clear picture of the effect of urbanization.

Pitfall trapping with a neutral preservative or dry traps, the method used for carabids, is a relative cost-effective, low-impact and time efficient method of sampling the ground foraging arthropod community (Spence and Niemela 1994), and despite concerns about the bias implicit in different sampling methods, remains the most suitable means of sampling diverse ground arthropods (Ausden 1996; in McIntyre et al. 2001). The taxa for which pitfall sampling is recommended are spiders (Arachnida), rove beetles (Coleoptera: Staphylinidae), ants (Hymenopterans), and certain plant and leaf hoppers (Hemiptera: Homoptera) (Work et al. 2002). Though the inclusion of these groups in addition to carabids may increase information available to researchers, there are practical issues with utilizing certain ground-dwelling groups. Spiders have relatively soft bodies and are difficult to identify; the necessity of extracting genitalia for identification pose problems to a researcher with limited time. Plant hoppers often have several nymphal stages, making identification difficult and abundance calculations tricky. Rove beetles are extremely difficult to identify, even by trained taxonomists (Peck and Thayer 2003), and may not offer particular insights into the arthropod assemblage without species-specific information as this taxon, like carabid beetles, has a wide variation in feeding habitats including mycovores and carnivores (Newton et al. 2001).

Beetles are relatively easy to identify to taxonomic families. Though families outside of the rove and carabids are incidental in pitfall traps and are not likely to represent the true diversity of their respective family assemblage, their function can point to important aspects of the habitat. Finally, as the most numerous order of animals on the planet, beetles occupy almost every trophic level of the system (Marshall 2006).

In this study we used a sub-sample of the order Coleoptera (those that fall into pitfall traps) to ask the following questions about the built environment: Do biodiversity measures differ in response to habitat type in urban areas, and if so, are these changes observed universally among groups? Does the addition of ecological information, such as the family feeding guild, add to the understanding of the habitat? What does data from different taxonomic levels (family, species) and or breadth (select beetles vs. all beetles) reveals about the urban environment?

For arthropods, and perhaps especially beetles, the species level of identification is impractical for planners and environmental consultants who need alternative methods to measure the taxonomically complex fauna (McIntyre et al. 2001). In order to investigate the tradeoffs of using species level versus family level identifications, and a higher (order) versus a lower (family) level of organization, we analyzed the results using three different groupings of beetle identification: 1) the family level (that is feasible for individuals with less expertise or time), 2) the species level (morphospecies), (for those interested in exploring in greater detail differences in habitat type or looking to ecological factors such as regional diversity), and 3) a subsample of the beetle data including only the carabid and rove beetle families, as these are two beetle taxon for which pitfall trapping is a recommended method.

Methods

Site Selection

The study was conducted during the summer of 2007 in Detroit, Michigan (42°19'53.76"N, 83°2'51"W, approximately 183m in elevation). Habitat types were represented as four replicates each of urban forests, vacant lots, and gardens for a total of twelve sites (Table 1, Site Descriptions; see Chapter II, Figure 1 for Map). Environmental characteristics for each site, including vegetation, land cover quantification, and weather

during sampling periods as recommended by McIntyre (2000), are provided in Table 2 and 3. Remnant forests sites were selected with the assistance of former city naturalist Susan Campbell, based on the age of the tree stands, floristic quality, and suitability for study (June 2006).

Each vacant lot was confirmed as a former residential or non-industrial commercial parcel through exploration of former tax records and zoning maps (City of Detroit Finance Department 2006; City of Detroit Planning and Development Department 2006). The lots were free of a structure for at least fifteen years prior to the study according to historical aerial photographs (USGS and USDA 1949-1997) and Sanborn Fire Insurance Maps (1992); this time frame allows the plant and insect communities respond to the disturbance of structural clearing and continual mowing (Gilbert 1989). No illegal dumping was observed in reviewed aerial photographs of these vacant lot sites, decreasing the probability of soil contamination that might alter the community structure. Urban garden managers volunteered in response to a request made to participants in the Detroit Agriculture Network. Each garden was managed without chemical pesticide use, though other cultural methods varied.

Carabid Sampling

Three pitfall traps consisting of two nested 0.473 liters (16 oz.) plastic tubs, 11.4 cm in diameter and 7.6 cm in depth, were placed 5 m apart along two short transects. The transects were placed 5 m apart, establishing a rectangle of six traps at each site (Chapter II, Figure 2 and 3). The tubs were sealed with a lid when not in use. This diameter of pitfall trap is unlikely to capture the largest invertebrate species or rare species, but should characterize the dominant fauna within the ground layer without the mammalian and amphibian by-catch of more inclusive larger traps (Work et al. 2002).

The traps were opened and filled with approximately 200 ml saturated saline solution with unscented detergent to break surface tension during 15-18 May, 16-19 June, 16-19 July, and 15-18 August of 2007. Saline solution has little attractive or repulsive characteristics, but preserves important morphological components of insects, making it the preferred preservative in passive sampling (Perner and Schueler 2004; Sasakawa 2007). A plastic plate supported with three nails served as a roof to exclude rainwater

and non-target taxa, positioned approximately 2-3 inches above each open trap during sampling.

The plot size was kept the same across sites in order to accommodate the smallest of sites (G4) though some trap-to-trap interference was likely (Digweed, Currie et al. 1995). The forest traps were placed in areas away from the forest edges to avoid edge effects, and at relatively higher elevations within the site due to seasonally flooded during the first sampling date in May and the water table would otherwise push the cups from the ground. Garden traps were placed near or among row plantings, but on request of the gardeners care was taken to choose a layout that would least disturb cultivated plants. All gardens were tilled approximately one to two days prior to the trap installation. Vacant lot traps were placed towards the center of the ruderal vacant area to avoid edge effects, though an attempt was made to place traps near occasional vegetation, such as trees or shrubs, in order to detect carabid species with plant structure preferences that may be present (Brose 2003).

Specimen cups were retrieved from the pitfall trap and contents were filtered with deionized water to remove salt solution, and then refrigerated in a 70% ethyl alcohol solution. Beetles were first sorted by order, family and then morphospecies following White and Peterson (1998) and Marshall (2006). Each family was determined as either detritovore, herbivore, mycovore, omnivore or predator when possible, using Arnett and Thomas (2000) and Watson and Dallwitz (2003-Present).

Representative individuals of each morphospecies were pinned when possible; voucher individuals of minute or delicate species were stored in 70% ethanol. Information regarding the pitfall station, site, habitat type, and sampling date was maintained with each individual.

Habitat Characterization

Environmental characteristics of each site were measured at three spatial scales (1 ha, 400 m², and 1 m²) to examine the factors that may correlate with beetle diversity and abundance. At each site 100 x 100 m plot (1 ha) and a 20 x 20 m plot (400 m²) were established, centered on the middle the baiting areas. In each 100 x 100 m plot the researcher and field assistant estimated percent of ground covered by a) bare ground, b)

understory vegetation (e.g. forbs, grasses, etc.), c) shrubs, d) concrete, and e) buildings, based on a system standardized between field investigators. The number of trees >30 cm circumference at breast height (cbh) was also counted.

Data on trees, shrubs and canopy cover was collected within 20 x 20 m plots. For trees a) counted and identified all trees >30 cm cbh, b) measured circumference at 1.37 m above the ground, and c) estimated height. For shrubs and tree seedlings, we a) counted and identified all stems < 1 m height, b) measured circumference at the ground, and c) measured height. The total richness of woody plants at each site was calculated by combining tree and shrub data. The percent canopy cover for each 20 x 20 m plot was estimated with a concave vertical densiometer. We took densiometer readings at the center and the four cardinal corners of the plot at the start of each pitfall sampling, and calculated mean values for each site over the season.

To establish the 1 m² plots, the 20 x 20 m plot were divided into 4 sections of 5 x 20 m. Within each section we randomly placed a 1 x 1 m plot using a generated random number chart to measure percent covered by a) bare ground, b) grasses, c) forbs/herbs, d) rocks/wood panels, e) leaf litter, and f) fallen branches. For each 1 x 1 m plot, we also recorded a) the height of the tallest non-woody plant, b) number of individuals of forbs/herbs, and c) number of species of forbs/herbs and grasses. The habitat data were collected once at each site between May-September 2007 in Detroit.

Analysis

Family and Species Richness

Family richness of all beetles, morphospecies richness of all beetles, and morphospecies richness of rove and carabid beetles were analyzed using species-accumulation curves for each habitat type with respective 95% confidence intervals constructed using the software program EstimateS included (Colwell 2005). Species richness estimates were not included because the observed accumulation curves did not suggest an asymptote for either the forest or vacant lot species, indicating that the estimates are unreliable.

Number of Beetles Captured

As the rate of capture of individuals in pitfall traps is associated with activity level and local density of the capture, the number of individuals captured at the site was standardized by dividing the individuals captured by the total number of traps, a measure referred to as “activity-density” (Thiele 1977). This also corrects for unequal sampling units when pitfall traps are disturbed. As traps in close proximity may have interactions within the same site (Digweed et al. 1995), and most sites are heterogeneous at the microsite scale, it is appropriate to consider each trap as a subsample of the site area (Greenslade 1964; Spence and Niemela 1994). The activity-density was calculated as the mean number of beetles captured per trap per three-day sampling period, and data was also pooled for the four-month sampling period at each site.

Activity-density measurements at the species and morphospecies were transformed prior to analysis using the $\text{SQRT}(x + .05)$ to normalize count data that follows a Poisson distribution and to equalize variance among the sites. A repeated-measures multivariate analysis of variance (MANOVA, or general linear model of repeated measures) followed by Tukey’s honestly significant differences (HSD) was used to compare the average morphospecies activity-density and richness of beetles between months and habitats. Tukey’s HSD corrects for the uneven sample number due to the missing May data for F1. A univariate ANOVA and Tukey’s HSD was then used to determine differences among the combined rove and carabid beetles by habitat. The entire morphospecies set (183 morphospecies) was too large for the SPSS software, so a univariate ANOVA using only the 88 morphospecies with more than one individual was performed to determine significant differences of morphospecies between sites.

Following the method of McIntyre et al. (McIntyre et al. 2001), beetle families were assessed for each site on a monthly basis, and then pooled for comparison with no activity-density transformations applied. A repeated-measures MANOVA followed by Tukey’s HSD was used to compare the average family richness between months and habitats. A univariate ANOVA was then used to determine differences in the number of families found at each site. A univariate ANOVA and Tukey’s HSD was used to determine differences among each family by habitat.

Shared Characteristics between Sites, Habitats

Hierarchical cluster analysis using the Bray-Curtis index for abundance data (Bray and Curtis 1957) based on the unweighted pair-group average (UPGMA) linking method of PAST v1.88 (Hammer et al. 1995-2008) was employed to create dendrograms representing the taxonomic similarity of each of the sites based on the three different beetle groupings and the habitat characteristics. This index has been shown to reflect the actual overlap accurately for virtual any underlying distribution, and is symmetrical for overlapping blocks – therefore, comparing the grouping and distance outcomes among the total beetle morphospecies, rove and carabid, and family groups applied to this study was feasible (Bloom 1981). The Bray-Curtis index for between-site and between-habitat comparisons of species richness was computed using EstimateS. A one-way analysis of similarity (ANOSIM) was also conducted using PAST v. 1.88 to determine significant differences between habitats based on a non-parametric test of the Bray-Curtis index values.

Climactic Variables

Climactic variables also influence the capture rate of arthropods (McIntyre 2000; Greenslade 1964; Thiele 1977; Frankie and Ehler 1978). In order to better understand the effects of climate, the average high and low temperatures and daily precipitation were calculated from data provided by The Weather Exchange network (2007). Single linear regressions comparing three taxonomic groupings, individuals, species, and families versus the average low temperature, high temperature, and precipitation were performed in order to look for trends in the capture rates and the climatic variables.

Habitat Characteristics

Hierarchical cluster analysis using the Bray-Curtis index (Bray and Curtis 1957) based on the unweighted pair-group average (UPGMA) linking method of PAST v1.88 (Hammer et al. 1995-2008) was employed to create a dendrogram representing the similarity among sites and between habitat types, expecting that sites of a habitat will be grouped more closely. A univariate ANOVA and Tukey's HSD was then used to determine differences among habitat characteristics by land-use. Percentage measurements were arcsine-transformed prior to analysis.

Results

Morphospecies richness for all beetles

A total of 183 morphospecies and 1039 individuals were captured in this study (Appendix X). 122 species and 752 individuals were captured in the gardens, 48 species and 133 individuals were captured in the vacant lots, and 49 species and 154 individuals in the forests. Figure 1 shows the average number of species and individuals for each of the habitats. As can be seen from this graph, the gardens habitat harbored the highest number of species and individuals, with the forest and vacant lot habitats not differing very much from one another. Differences were not found in the overall activity-density by month utilizing the multivariate repeated measure analysis; however, significant differences in the variance of activity-density existed for the June, July and August sampling periods ($F = 14.474$, $df = 2, 9$, $p=0.002$, $F= 13.734$, $df = 2, 9$, $p=0.002$ and $F = 8.322$, $df = 2, 9$, $p=0.009$, respectively), with the average activity-density of the gardens in generally exceeding that of the other habitats. The average number of species captured also differed seasonally, with more species captured in the gardens than in the vacant lots in July ($F = 4.841$, $df = 2, 9$, $p = 0.037$) and in the gardens than either of the other habitats in August ($F = 8.816$, $df = 2, 9$, $p = 0.008$). Despite of these differences, the species accumulation curves show no significant differences in cumulative species richness among the three sites (Figure 2a). It is obvious in this Figure that the garden capture had significantly more individuals than the other two sites. A truncation of these curves at 135 individual indicates no significant differences in species richness among sites ($p > 0.05$; Figure 2b).

Of the 183 species, only 2 of these species were found in all three habitats, a Nitidulidae (sap beetle) and a Lathridiidae (minute brown scavenger beetle). Excluding these 2 species, there are 27 species shared between gardens and vacant lots, 5 species shared between gardens and forests, and 2 shared species between forests and vacant lots. This accounts for 19.67% of total species, with the remainder occurring only a single habitat type.

Within the habitat types, the majority of species were found only in a single site. The gardens had the greatest percentage of shared species, with 44, or 36.07% of the 122

species found in multiple sites. Of the 48 species captured in vacant lots, 12 were found in multiple sites, accounting for 25% of the total. Among the forests, 11 of the 49 species, or 22.45%, were found in more than one site. The cluster analysis shows the three habitats types branching relatively early, and in general sites are grouped by habitat type. Forest sites 2, 3 and 4 share the highest Bray-Curtis indices (Table 5). The one site that appears distant from the others of its habitat type is G4, which is most similar to the vacant lots (Figure 3). ANOSIM values (Figure 3) indicate significant differences between the habitat types when grouped by morphospecies with the greatest similarity between gardens and forests ($R = 0.9583$, $p = 0.0276$).

Of the 88 morphospecies with more than one individual captured, only four were associated with a particular habitat site. A dung beetle (family Aphodiine) species ($F = 7.737$, $df = 2, 9$, $p = 0.11$) and a round fungus beetle species (family Leodidae) ($F = 9.000$, $df = 2, 9$, $p = 0.007$) were found primarily in gardens. A species of minute brown scavenger beetle (family Lathridiidae) ($F = 5.609$, $df = 2, 9$, $p = 0.026$) was found in abundance in gardens and vacant lots, but was absent in the forests. The ground beetle *Poecilus lucubladus* (*lucubladus*) ($F = 9.484$, $df = 2, 9$, $p = 0.006$) was found in all forest sites, but not in gardens or vacant lots.

Richness and Abundance at the Family Level

A total of 27 beetle families were captured during the study: 22 in the gardens, 13 in the vacant lots and 10 in the forests (Table 5). The average number of families was significantly higher in the gardens (14 ± 4.24) followed by the vacant lots (8.75 ± 0.50) and the forests (5 ± 0.82) ($df = 2, 9$, $F = 12.965$, $p = 0.002$) (Figure 4). The average abundance of families also differed significantly, with gardens significantly higher in July ($F = 7.426$, $df = 2, 9$, $p = 0.012$) and August ($F = 8.726$, $df = 2, 9$, $p = 0.008$). During these later two months, the gardens differed from the both the vacant lots ($p = 0.024$ and $p = 0.036$) and the forests ($p = 0.019$ and $p = 0.008$, respective to month). Accumulation curves at the family level show no significant differences among habitats for family richness (Figure 5), although it is evident that the curve for the vacant lot is reaching an asymptote while those for the forest and gardens continue to rise.

Almost half of the families, 13 of 27, were represented in more than one of the habitats (Table 5). Seven of the families, or just over a quarter, were collected exclusively in the gardens and vacant lots, with no representatives in the forests. No families were shared by the forest and garden habitats alone, though about one fifth, or 6 of the 27 families were found in all three habitats. The cluster analysis of the family taxon (Figure 6) shows G1, G2 and G3 clustering together, least similar from the other sites. The four vacant lots all cluster with one another, with V1 and V4 showing the most similar family characteristics among this habitat. G4 branches just before the forests, indicating that it shares more family similarities with the forests than with either the other gardens or vacant lots. The forests appear more similar among sites than either of the other habitats, an observation supported by the Bray-Cutis Index values (Table 4).

Figure 7 shows the distribution of individuals per beetle family for all 12 sampling sites. Both the garden plots and the vacant lot plots had more families represented than the forest plots. Examining the five most abundant families of beetles at each site and their percentage of the total beetle capture, the dominance of certain families is noticeable for the garden and forest sites, more than for the vacant lots (Figure 8). Of the 751 individuals captured in garden habitats, 41.41% were rove beetles (Staphylinidae), 18.91% were leaf beetles (Chrysomelidae), 10.52% were ground beetles (Carabidae), 6.13% were weevils (Curculionidae), and 5.46% were silken fungus beetles (Cryptophagidae). Finally, of the 154 individuals captured in the forest, 69.48% were carabid beetles, 9.74% were rove beetles, 9.09% were sap beetles (Nitidulidae) and 7.14% were weevils; all other families were singletons. Of the 133 beetles found in vacant lots, 21.80% were weevils, 19.55% were minute brown scavenger beetles (Lathridiidae), 14.29% were rove beetles, 9.77% were carabid beetles, and 9.02% were leaf beetles. G4 is dominated by the ground beetles, accounting for 30.86% of individuals, followed by the rove beetles (25.93%) and then weevils (17.28%), a distribution that is more similar to the forests for the lack of leaf beetles or other dominant garden families, as represented in the cluster analysis (Figure 6). ANOSIM values (Figure 6) indicate significant differences between the habitat types when grouped by family, with the strongest similarity between forests and vacant lots ($R = 0.9792$, $p = 0.0264$).

Several of the families differed in their distribution among the sites. The ground beetles ($df = 2, 9, F = 7.527, p=0.012$), weevils ($df = 2, 9, F = 4.580, p=0.042$), minute brown scavenger beetles ($df = 2, 9, F = 6.012, p=0.022$), round fungus beetles (Leiodidae) ($df = 2, 9, F = 9.000, p=0.007$), root-eating beetles [a misnomer] (Monotomidae) ($df = 2, 9, F = 6.750, p=0.039$), and scarab beetle (Scarabaeidae) ($df = 2, 9, F = 5.583, p=0.030$) differed in respect to their average abundance in the sites, as indicated by the ANOVA test comparing the abundance of each family at each site. Tukey's post-hoc tests indicated that the forest ground beetles outnumbered the vacant lot ground beetles by more than 8:1 ($n = 107$ and 13 , respectively, $p=0.011$). The weevils were more than four times greater in abundance in the gardens than in the forests ($n= 46$ and 11 , $p=0.035$). Minute brown scavenger beetles were primarily found in the vacant lots, especially compared to forests where only a single individual was captured ($n = 26$ and 1 , $p=0.027$). Scarab beetles were most numerous in the gardens while none were found in the forests ($n = 9, 0 p=0.031$). Though differences were found for the round fungus beetles ($n=3, p = 0.13$), this family was found only in the gardens, leading to a rather obvious significance between the gardens and other habitat types.

Ground and Rove Beetles

Two families of beetles, the ground and rove beetles, comprised the majority (52.36%) of the 1039 beetles captured in this study. A total of 199 ground beetles and 345 rove beetles, comprising 30 species of ground beetles and 46 species of rove beetles, account for of 41.53% of the total species identified (Figure 9). The overlap of the confidence intervals of the species-area curve for the rove and carabid beetles shows no significant difference between the species richness of the gardens, vacant lots, and forests (Figure 10). Though 60 of these species were captured in only one habitat, only the rove beetle morphospecies "Aleocharinae sp.2", found primarily in the gardens, and *Poecilus lucubladus* (lucubladus), found only in the forests, differed significantly between the habitat types ($df = 2, 9, F = 5.802, p = 0.024$ and $df = 2, 9, F = 16.040, p = 0.001$) due to individuals being present in each site of the respective habitat type.

The greatest number of shared species was observed between G1 and G2, which share 11 species out of the 23 ground and rove beetle species found at each site. Six

species are shared among the site pairs of G2 and G3, and G1 and V3. The cluster analysis for this grouping of beetles shows several branch points at low Bray-Curtis index numbers, indicating comparatively less similarity between the sites within a habitat than the family representation, and similar to that seen in the total morphospecies dendrogram (Figure 12, Figure 6, Figure 3). Though the forest, vacant lots, and gardens are still primarily grouped together, two sites appear on branches further from their habitat types. V2 appears to be more closely related to the forest sites than to the gardens, and in general is the first branch point at zero. G4 appears to be more similar to the vacant lot sites than the other garden sites. ANOSIM values (Figure 12) indicate significant differences between the gardens and forests, and forests and vacant lot habitat types when grouped by ground and rove beetle species, with the greatest similarity between gardens and forests ($R = 0.9792$, $p = 0.0283$).

Feeding guilds

The predominant feeding guild in the garden and forest sites appears to be carnivorous beetles, while the vacant lots are more equitably distributed among the guilds (Figure 9). The herbivore ($df = 2, 9$, $F = 7.772$, $p=0.011$) and carnivore ($df = 2, 9$, $F = 9.021$, $p=0.007$) feeding guilds differed significantly with respect to habitat. Significantly more herbivores were captured in gardens than either the vacant lots ($p=0.039$) or the forests ($p=0.012$), and significantly more carnivores were found in the gardens than in the vacant lots ($p=0.006$) (Figure 10). Mycovores were more abundant in the gardens, and were on the significantly more abundant than those found in the forests ($p = 0.045$).

Habitat Characteristics

Habitat characteristic analysis illustrates that the habitats are indeed significantly different in environmental factors. The cluster analysis of habitat characteristics (Figure 15) shows little similarity between the remnant forests and the other two land uses of gardens and vacant lots. Of the forests, F1 appears to be the least similar, with F2 and F4 being the most similar. V4 is the least similar of the other vacant lots, with V1 and V3

appearing to be the most similar. G3 differs the most from the other gardens, with G1 and G2 being the most similar.

The results of the Tukey's HSD following and ANOVA comparing the listed characteristics by site (Table 6) shows that several site characteristics were significantly different between habitat types. In general forests are dominated by trees, shrubs and canopy cover, while vacant lots are dominated by grass and gardens by forbs and open areas. The gardens and vacant lots had significantly more impermeable areas in the hectare area surrounding the sampling site. Based on the findings in Table 6, gardens and forests differed the most significantly based on the habitat factors considered, while vacant lots had a few intermediate values that were similar to the forests.

Climate Factors

Diversity factors appear to be affected by climatic events. Greater precipitation decreased the number of individuals and species that were captured (Figure 16 a), while increasing high and low temperatures resulted in higher numbers of species and individuals being captured (Figure 16 b and Figure 16 c). However, the number of families did not appear to be as affected by changing climactic factors.

Discussion

The variety of land uses is one of the distinguishing characteristics of the urban matrix. Though categorizing land by its use is somewhat descriptive, the actual structure or function of the habitat can vary within each of these uses. Figure 15 shows that though the sites do cluster by habitat, these clusters do have variance. When the habitat characteristics are analyzed (Table 6), the significant differences between habitats are revealed. The cluster analysis using different biological measures (all beetles by morphospecies, all beetles by family, and all the carabid and rove beetles by species) the resulting dendrograms are affected by different aspects of the beetle assemblages. For instance, the beetle morphospecies cluster (Figure 3) shows that G4 is more similar to the vacant lots than to the other gardens. Considering that G4 is the smallest of our gardens, and is surrounded on two sides by grassy lots, this may indicate that the site is colonized or frequently visited by surrounding beetle species, or that the sampling edge effect is

particularly strong at this site as traps always lay close to the garden boundary. However, if the beetle families are used in cluster analysis (Figure 6), then G4 moves from the vacant lot cluster to the forest cluster. This is reasonable, considering that carabid beetles dominated the forests, as they did G4. If the analysis was limited to rove and carabid beetles only, then V2 would be considered quite different from all the other sites due to the presence of one unique carabid individual that is shared with F3. Here, G4 would be most similar to the vacant lots that share similar rove beetle species (Figure 12), instead of the forests that do not share the same species of carabids (see Chapter II). This seems to be an ecologically valid association considering G4's proximity to vast vacant lot areas.

The physical distance between sites does not appear to be a factor in the shared species of the rove and carabid beetle grouping; for instance, G1 and V3 share more species than G1 and V4 (Chapter II, Figure 1). The differences in cluster analysis outcome, confirms that the biological measure does indeed change the perception of the site similarity. The fact that G4, the smallest study site, appears to change most significantly depending on the grouping suggests that small parcels may contain less predictable assemblages. The influence of environmental characteristics immediately surrounding the habitat patch appears to have a greater influence on the beetle community when the site is small, as noted by the rove and carabid beetle cluster analysis.

It is clear that the gardens contain many more beetle families and individuals than the other habitats (Figure 1 and Figure 4), and the species area curves (Figure 2 and Figure 5) did seem to indicate that an asymptote had been reached in the case of species and families in this habitat. The conservative Jackknife1 species estimator estimates that 182 species, 60 more than the observed 122 species, are likely to exist in the gardens, with other estimators predicting higher values (Colwell 2005). The species-area curves revealed no significant differences in the species or family richness among the habitats (Figure 2, Figure 5 and Figure 10), regardless of the taxon grouping. However, the beetle families (Figure 5) reveals that this taxon grouping comes closest to reaching asymptotes in all habitat types when compared to the other taxonomic groupings.

Though carnivorous species appear to dominate most garden and forest sites (Figure 13), this grouping may be deceptive because two of the most numerous families, the Carabidae and the Staphylinidae, were included with the carnivores despite their more varied eating habits, as predation is considered the primary feeding preference these families. Chapter 2 explores the eating habits of the specific species of carabid beetles, and finds that several of carabid species found in the gardens are partially or primarily herbivorous. Despite this complexity, it is likely that the forests and gardens foster food resources, such as slugs, larvae and aphids that serve as good prey for the carabid beetles. In addition to having more herbivores than either other habitat, and more predators than the vacant lots, the gardens held the majority of the mycovores. This indicates that the breakdown of organic matter is ongoing at the garden sites, and is likely a result of the organic amendments applied to the soil or the regular exposure of organic matter through tilling.

The gardens were dominated by the rove beetles, especially in G1 and G2, which were the largest and appeared to be the most diverse in microsites of the gardens in the study. G2 also had the most carabid beetles out of the gardens, primarily consisting of *Scarites subterraneus*. This beetle may survive especially well in spite of disturbance of garden soil because it lays its eggs off the ground, attached to leaves by mud (Larochelle and Larivere 2003). See chapter 2 for more on the carabid beetles of this study. The ladybird beetles (Coccinellidae) were also found more frequently in gardens, likely due to the presence of attractive prey such as aphids.

Like the rove beetles, the weevils are a very diverse family of beetles, but are known for their destructive abilities on plants and seeds. Though they are found in all habitats, gardens and vacant lots fostered the greatest numbers of weevils. However, the species also varied by habitat, and specialist weevils such as the Clover Curculio were identified in the gardens. High amounts of seeds are available at certain times of the year from the grasses and ruderal forbs in vacant lots, and thus the presence of weevils is not surprising.

The leaf beetles (Chrysomelidae), the primary issue for the urban gardeners, were primarily garden-dwellers. Most of these species were flea or squash beetles; pests of horticultural crops. Higher quality plants – those under the least nutrient, water or light

stress – are generally more attractive to pests; (Andow 1991), which in addition to the sweeter or less toxic leaves that accompany edible crops, may be why leaf beetles are so abundant in the gardens. Two alternative hypothesis also exist: 1) as garden plants are relatively rare in the urban environment, they represent a scarce resource that is particularly attractive to herbaceous beetles (Hanks and Denno 1993); 2) as many of the garden plants in Detroit are under water stress, their concentrated plant sap is a higher quality resource than the naturally occurring forbs (in McIntyre 2000; Schmitz 1996).

The Oedemeridae are of special interest; this family is represented by a single species, the *Nacorderdes melanura*, commonly known as the Wharf Borer. This species is herbaceous as an adult, usually on the flowers of Queen Anne's Lace, but feeds off of decaying wharfs and damp rotting wood as larvae. Both the beetle and the plant are European in origin. These beetles were found in G2, approximately 1.75 km from the shorefront of the Detroit River, but the actual source of the beetles is not known.

Seasonal variation in the beetle assemblages does indeed occur. The activity-density of all captured beetles increased throughout the season in all habitats, peaking in July and only slightly declining in August. May was the only month in which the three habitats were not significantly different in total beetle activity-density, likely because of the low capture rates in the flooded forests and relative barren nature of the gardens, which in most cases had just finished tilling and were beginning the growing season. Many arthropods may have not yet begun their adult life stage at this point, as only 23 degree days (base 50) had occurred (Weather Exchange 2007), and heavy precipitation may have further constrained activity (Figure 16a). There were more individuals, species and families of beetles captured by pitfalls in the gardens than either of the other habitat types after the month of May. The seasonal growth of plants attractive to herbivorous beetles and other animals such as aphids that can then become prey is a likely cause.

The seemingly low number of families in forests is somewhat puzzling, considering that this habitat appears to be the least impacted by urbanization. A simple technological answer - that the pitfall traps do not capture the beetles dwelling in the overstory canopy and small tree/shrub layer of the forests. The seasonal nature of the forest flooding and patchy native flora likely limits the species of herbaceous-dwelling beetles. However, the larger predaceous carabid beetles appear to dominate the forest

floor (Figure 9, see also Chapter 2), as they can consume small invertebrates such as worms and slugs throughout the season. In contrast, the large areas of dense grassy vegetation found in the vacant lots may block beetle movement and reduce the capture rate (Luff 1975) (Table 6).

When considering the beetle families individually (Table 5), it is clear that some families were exclusive to certain habitat types, or restricted to just two of the three habitats, while others were generalists. In the forests, several families were represented by a single individual, which were likely incidental captures of beetles that live higher in the vegetational column, or were otherwise not frequently captured by pitfall traps. These include the families Bupresitidae, Cantharidae, Cerophytidae, Lampyridae, and Trogidae. Besides the Carabidae, only the Nititulidae were also found in equitable numbers in the forests and gardens – but the species diversity in the forests was higher for this family. The gardens also had two unique families represented by one individual – the Phlacridae and the Ptiliidae. Of the twenty remaining families, seven were present exclusive to the gardens: Anthicidae, Biphyllidae, Leiodidae, Monotomidae, Mycetophagidae, Oedemeridae, and Throscidae. The majority of the garden beetle families mentioned eat fungal matter or otherwise play a role in degrading organic matter, which may have either been fostered by, or imported with organic materials such as compost that were brought in to amend the soil. Accordingly, the Corylophidae and Cryptophagidae, both mycophilous families, were primarily found in gardens, but also occurred in vacant lots. The omnivorous Elatridae, Histeridae, and Lathriidae were found in relatively equivalent abundance between the gardens and vacant lots. The Curculionidae were also found in equivalent numbers in the vacant lots and gardens, with a few in the forests as well, but the morphospecies appeared to be sorted by habitat type.

Conclusions

The environmental measures that characterize the habitats, along with the three groupings of beetles, tended to cluster the study sites in a similar manner. But different conclusions regarding individual sites could be drawn from analyses conducted at different taxonomic resolutions. The broadest and least specific grouping, the family taxon, succeeds in producing estimations of family richness present in the garden and

vacant lot habitats that are likely accurate. But when the feeding guilds are considered, this grouping fails to delineate significant differences in the beetle community structure because of generalizations made about families that have variable feeding habits - details about species with variable feeding habits are lost at the family resolution of analysis. This could lead to erroneous ideas that infer that a garden site is benefiting from the presence of the carabids, when those carabids are actually consuming plants, or that the prevalence of carabids in a garden may infer a resource value similar to a forest, when in fact none of the species are forest-dwelling carabids.

Considering all the morphospecies allows a finer resolution in the sorting of the data while breadth of analysis preserves all captured beetles. This cluster analysis more accurately characterizes these study sites. However, this measure cannot be considered accurate in estimating the actual number of beetle species in a habitat because many of the species are not appropriately sampled with pitfall trapping. This might lead a conservation biologist to undervalue habitats such as the forests or even vacant lots, where beetles species may be less ground-mobile due to alternate niches in the forests' canopy or the vacant lots' grass. Because most of a garden's vegetation is low to the ground, it may be that a morphospecies analysis more accurately characterizes this beetle community because incidental captures are more likely.

The study appears to confirm that it is most ecologically accurate to consider the species diversity of the two families of beetles that inhabit the soil surface – the carabid and rove beetles. This narrower scope of analysis still indicates a greater abundance of beetles in the garden settling (Figure 9), but provides a more nuanced look at species diversity. The species rarefaction curve suggests that though more species have been discovered in the gardens, the potential for the vacant lots and forests to harbor equivalent or greater diversity is possible (Figure 10). A more equivalent variance in the number of species that were observed suggests that the rove and carabid beetle grouping is a more accurate sampling method.

Within the literature of urban ecology, findings show gains and losses in species richness in association with urbanization, depending on the taxon, level of urbanization, and pre-existing environment (McKinney 2008). This study's examination of data in various urban habitats supports the idea that appropriate factors and taxon should be

taken into consideration when examining ecological responses to urbanization. For example, the vegetational cover and impermeable cover surrounding a garden could be similar, but the size of the garden, internal vegetational structure and site management may have a strong influence on the beetle diversity and abundance than the surrounding the external environment.

The high level of biodiversity in urban gardens, as well as the differences observed among garden taxa, suggests that gardens provide a unique and diverse habitat for ground-level beetles and possibly other fauna. As such, gardens are an accessible option for educators looking for opportunities for youth exploration of the sciences. For the academic world, agroecological studies aiming to increase beneficial insects or understand urban soil regeneration may be ripe topics of study.

Researchers and naturalists attempting to create a complete species list for urban conservation found two- to three-thousand invertebrate animals in the tree canopy of parks in Warsaw (Luniak 2008); while such information is interesting, it is an overwhelming and likely superfluous task for management. Pitfall studies of the rove and carabid beetles would cost little time and expense, and this study suggests that when combined with morphological characteristics of the carabids detailed in Chapter II, such a method may hold promise as a simplified study design for characterizing the conservation value of habitats.

| Table 1 | | | | |
|--------------------------|--|---------------------------|-----------------------------|---------------|
| Site Descriptions | | | | |
| Site Name | Site Location | Area m² | Lat/Long coordinates | |
| Garden 1 (G1) | Hope Takes Root, Wabash St at Perry St | 1830 | 42°20'9.14"N | 83° 4'40.78"W |
| Garden 2 (G2) | Earthworks, near 1264 Meldrum St | 2665 | 42°21'15.69"N | 83° 0'44.11"W |
| Garden 3 (G3) | Birdtown Garden, near 3443 Cass Ave | 828 | 42°20'40.50"N | 83° 3'36.72"W |
| Garden 4 (G4) | Acres of Hope, 5930 Woodward Ave | 353 | 42°21'50.51"N | 83° 4'8.64"W |
| Vacant lot 1 (V1) | 20336-20330 Lyndon St | 706 | 42°20'29.48"N | 82°58'28.92"W |
| Vacant lot 2 (V2) | 4174-4180 McClellan St | 306 | 42°25'7.16"N | 82°55'18.93"W |
| Vacant lot 3 (V3) | 8366 Thaddeus St | 306 | 42°20'51.30"N | 83°14'51.10"W |
| Vacant lot 4 (V4) | 5773 Wabash St | 520 | 42°25'27.66"N | 83° 6'55.30"W |
| Forest 1 (F1) | Belle Isle | 737500 | 42°23'36.78"N | 83°14'27.81"W |
| Forest 2 (F2) | Balduck Park | 28500 | 42°22'31.42"N | 82°59'45.74"W |
| Forest 3 (F3) | River Rouge Park | 368500 | 42°17'32.52"N | 83° 7'7.79"W |
| Forest 4 (F4) | Palmer Park | 258000 | 42°21'25.01"N | 83° 5'23.12"W |

Table 2. Average Values for Habitat Environmental Characteristics

| | Garden Avg. | Vacant Lot Avg. | Forest Avg. |
|--|--------------------|------------------------|--------------------|
| 1,000 m² (1 ha) | | | |
| % impermeable (concrete/asphalt) | 22.00 | 22.50 | 1.25 |
| % vegetation | 39.00 | 70.00 | 21.25 |
| % building area | 20.75 | 7.75 | 0 |
| % shrubs | 4.75 | 1.00 | 32.50 |
| % bare ground | 3.00 | 1.25 | 3.00 |
| # trees greater than 30cm in circumference | 30.75 | 45.50 | 321.75 |
| # buildings | 3.75 | 12.75 | 0 |
| 400 m² | | | |
| Avg Canopy Cover (%) | 5.41 | 10.79 | 92.54 |
| richness of woody species | 1.25 | 2.50 | 11.75 |
| tree density (3trees/400m2) | 0.00 | 0.01 | 0.03 |
| Avg tree height | 0.41 | 9.09 | 14.60 |
| Avg tree circumference | 7.75 | 106.52 | 111.21 |
| shrub density | 0.02 | 0.00 | 0.24 |
| Avg shrub height | 0.90 | 0.73 | 1.42 |
| Avg shrub circumference | 1.30 | 1.16 | 1.73 |
| 1 m² | | | |
| Average of % cover grasses | 21.19 | 81.60 | 1.88 |
| Average of % cover bare ground | 30.22 | 6.44 | 1.60 |
| Average of % cover forbs/herbs | 38.10 | 10.19 | 21.35 |
| Average of % cover rocks/wood panels | 1.38 | 1.63 | 6.25 |
| Average of % cover by leaf litter | 0.00 | 74.53 | 86.81 |
| Average of % cover by mulch | 8.88 | 0.00 | 0.00 |
| Average of % cover by fallen branches | 1.44 | 1.19 | 8.32 |
| Average of height of tallest non-woody vegetation (cm) | 29.06 | 38.00 | 28.63 |

Table 3. Temperature (°C) and Precipitation (cm) over Sampling Days (The Weather Exchange 2007)

| Date (2007) | Mean Temperature | High Temperature | Low Temperature | Precipitation |
|--------------------|-------------------------|-------------------------|------------------------|----------------------|
| May 15 | 22 | 30 | 12 | 1.07 |
| May 16 | 11 | 13 | 7 | 0.18 |
| May 17 | 10 | 12 | 5 | 0.13 |
| May 18 | 11 | 13 | 7 | 0.18 |
| June 16 | 22 | 28 | 15 | 0 |
| June 17 | 25 | 30 | 21 | 0 |
| June 18 | 26 | 33 | 21 | 0 |
| June 19 | 24 | 27 | 18 | 0.10 |
| July 16 | 22 | 27 | 16 | 0 |
| July 17 | 24 | 22 | 18 | 0.18 |
| July 18 | 24 | 30 | 18 | 0.03 |
| July 19 | 22 | 28 | 17 | 0.81 |
| August 15 | 23 | 26 | 21 | 0 |
| August 16 | 26 | 30 | 22 | 0 |
| August 17 | 22 | 27 | 17 | 0 |
| August 18 | 16 | 21 | 12 | 0.03 |

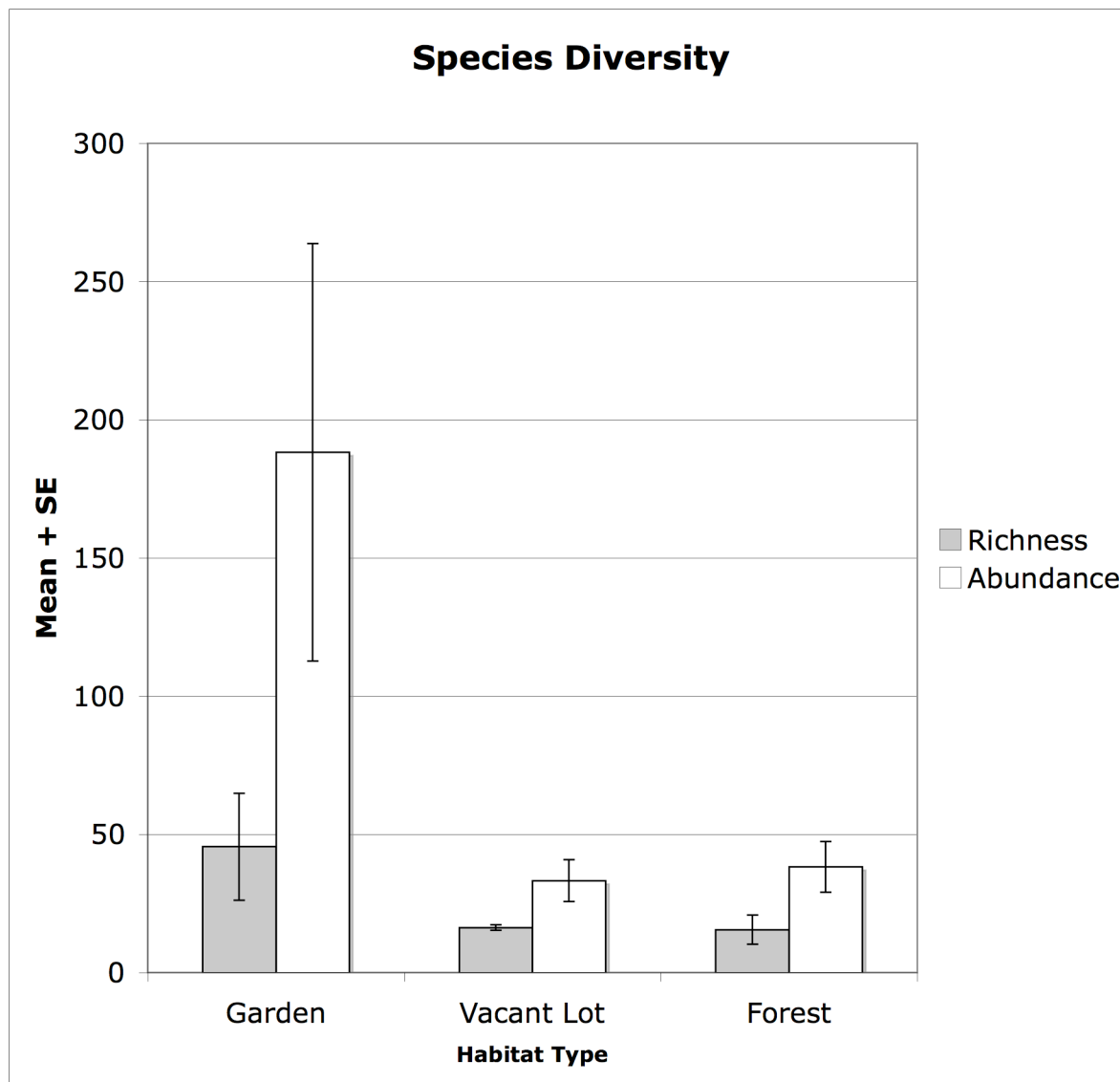


Figure 1. The average species richness and number of individuals captured from each habitat type

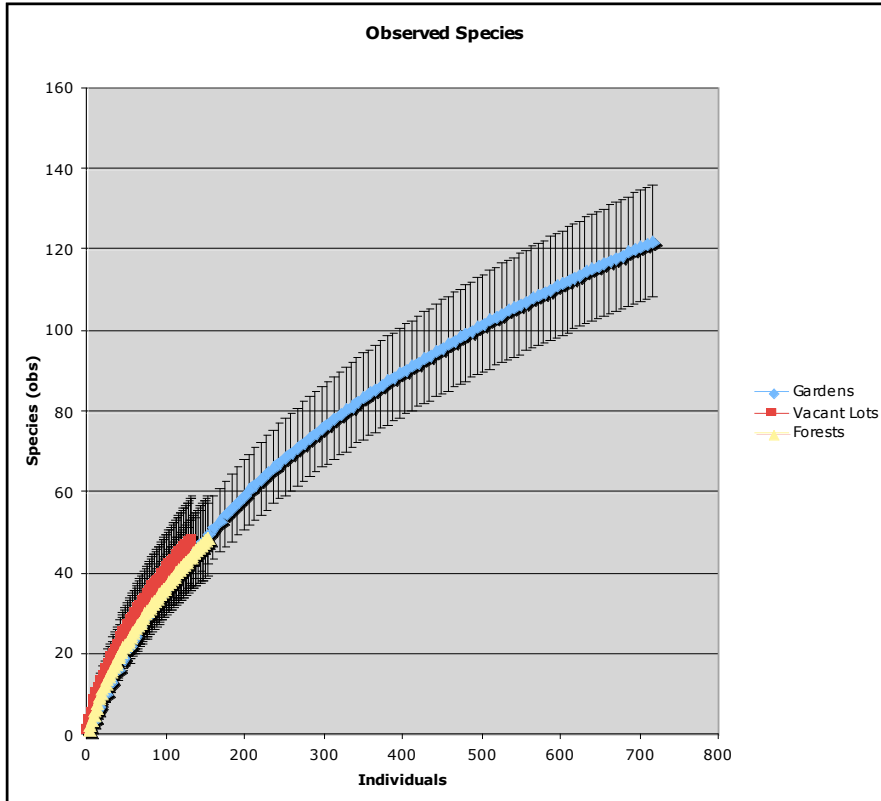


Figure 2a. The species rarefaction curves for all beetle species captured in the study.

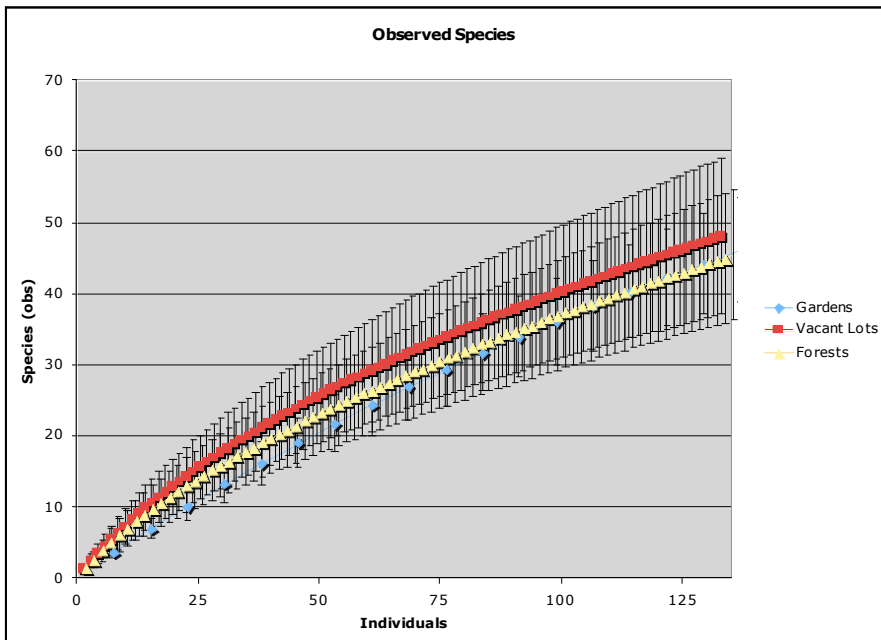


Figure 2b. An enlarged view of the curves, truncated at 135 individuals.

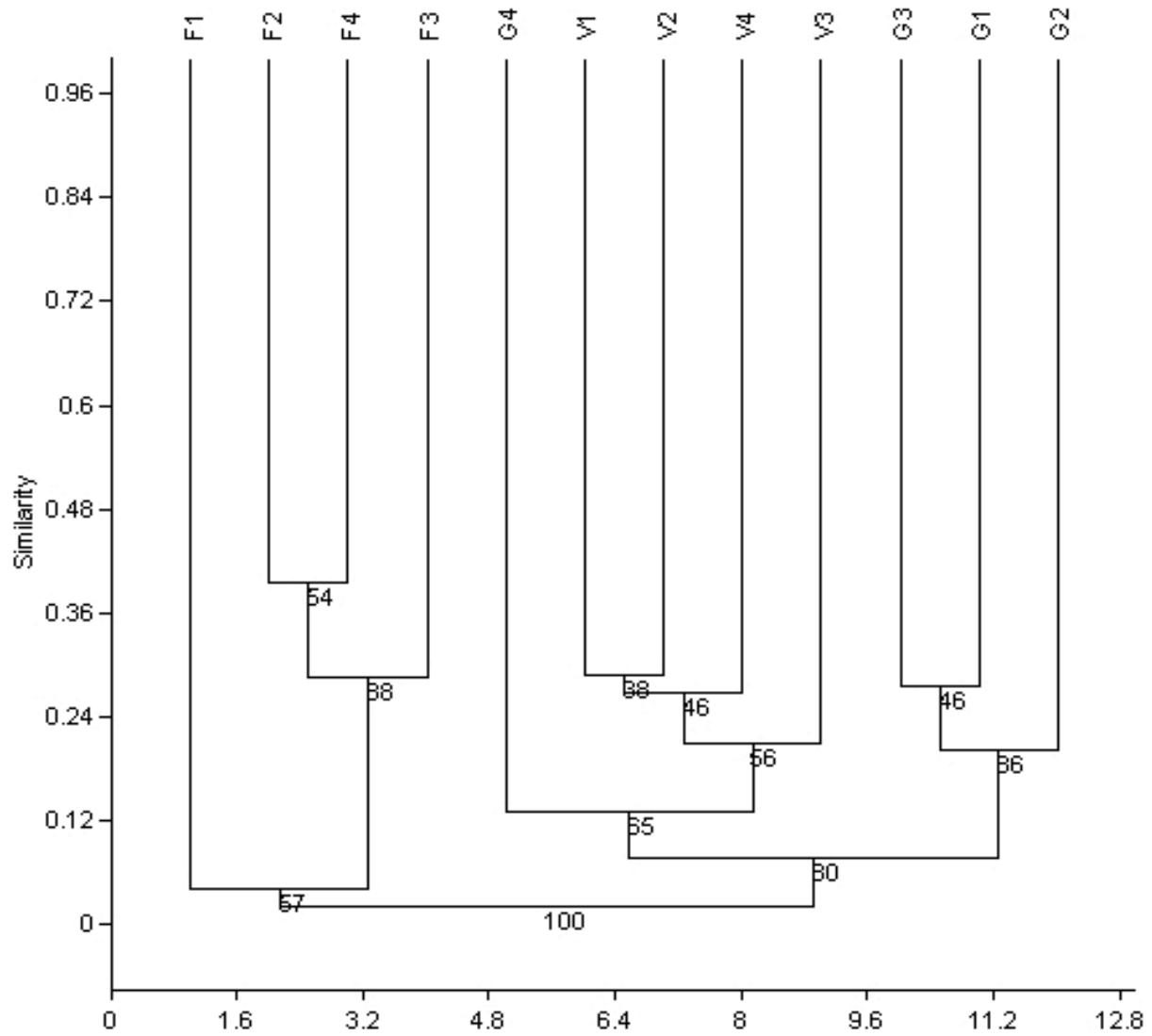


Figure 3. Bray-Curtis cluster analysis dendrogram for beetle morphospecies

| ANOSIM comparison | R-value | p-value |
|-------------------|---------|---------|
| G vs. F | 0.9583 | 0.0276 |
| G vs. V | 0.7188 | 0.0290 |
| F vs. V | 0.8570 | 0.0253 |

Table 4. Top Ten Similar Sites by Bray-Curtis Index

| Pairings | | | Bray-Curtis Index Value | | | |
|----------------|-----------|-----------------------|-------------------------|----------|------------------------|-------|
| Morpho-species | Families | Ground & Rove Beetles | Morpho-species | Families | Carabid & Rove Beetles | |
| F2 vs. F3 | F1 vs. F3 | G3 vs. V4 | | 0.200 | 0.564 | 0.136 |
| V2 vs. V3 | V1 vs. V2 | G4 vs. V3 | | 0.203 | 0.607 | 0.138 |
| V3 vs. V4 | F1 vs. F2 | G4 vs. V4 | | 0.234 | 0.613 | 0.140 |
| V2 vs. V4 | F2 vs. F3 | G3 vs. V3 | | 0.265 | 0.625 | 0.178 |
| G1 vs. G2 | G1 vs. G2 | G2 vs. G3 | | 0.268 | 0.627 | 0.225 |
| V1 vs. V4 | F3 vs. F4 | F2 vs. F3 | | 0.270 | 0.630 | 0.235 |
| G1 vs. G3 | G4 vs. F4 | G1 vs. G2 | | 0.274 | 0.640 | 0.289 |
| V1 vs. V2 | V1 vs. V4 | F2 vs. F4 | | 0.286 | 0.649 | 0.370 |
| F3 vs. F4 | F1 vs. F4 | V3 vs. V4 | | 0.370 | 0.676 | 0.435 |
| F2 vs. F4 | F2 vs. F4 | F3 vs. F4 | | 0.395 | 0.816 | 0.444 |

Table 5. Families of Beetles Captured During the Study

| Family | Feeding Guild | Garden | Vacant | Forest |
|----------------|---------------|--------|--------|--------|
| Anthicidae | Omnivore | 8 | 0 | 0 |
| Biphyllidae | Mycovore | 2 | 0 | 0 |
| Buprestidae | Herbivore | 0 | 0 | 1 |
| Byturidae | Herbivore | 7 | 8 | 0 |
| Cantharidae | Omnivore | 0 | 0 | 1 |
| Carabidae | Carnivore | 79 | 13 | 107 |
| Cerophytidae | Omnivore | 0 | 0 | 1 |
| Chrysomelidae | Herbivore | 142 | 12 | 1 |
| Coccinellidae | Carnivore | 4 | 1 | 0 |
| Corylophidae | Mycovore | 15 | 2 | 0 |
| Cryptophagidae | Mycovore | 41 | 7 | 0 |
| Curculionidae | Herbivore | 46 | 29 | 11 |
| Elateridae | Omnivore | 6 | 6 | 0 |
| Histeridae | Omnivore | 2 | 1 | 0 |
| Lampyridae | Carnivore | 0 | 0 | 1 |
| Lathridiidae | Omnivore | 20 | 26 | 1 |
| Leiodidae | Mycovore | 3 | 0 | 0 |
| Monotomidae | Detritovore | 9 | 0 | 0 |
| Mycetophagidae | Mycovore | 4 | 0 | 0 |
| Nitidulidae | Mycovore | 16 | 2 | 14 |
| Oedemeridae | Herbivore | 24 | 0 | 0 |
| Phalacridae | Herbivore | 1 | 0 | 0 |
| Ptiliidae | ? | 1 | 0 | 0 |
| Scarabaeidae | Detroitovore | 9 | 7 | 0 |
| Staphylinidae | ? | 311 | 19 | 15 |
| Throscidae | Carnivore | 2 | 0 | 0 |
| Trogidae | Detritovore | 0 | 0 | 1 |

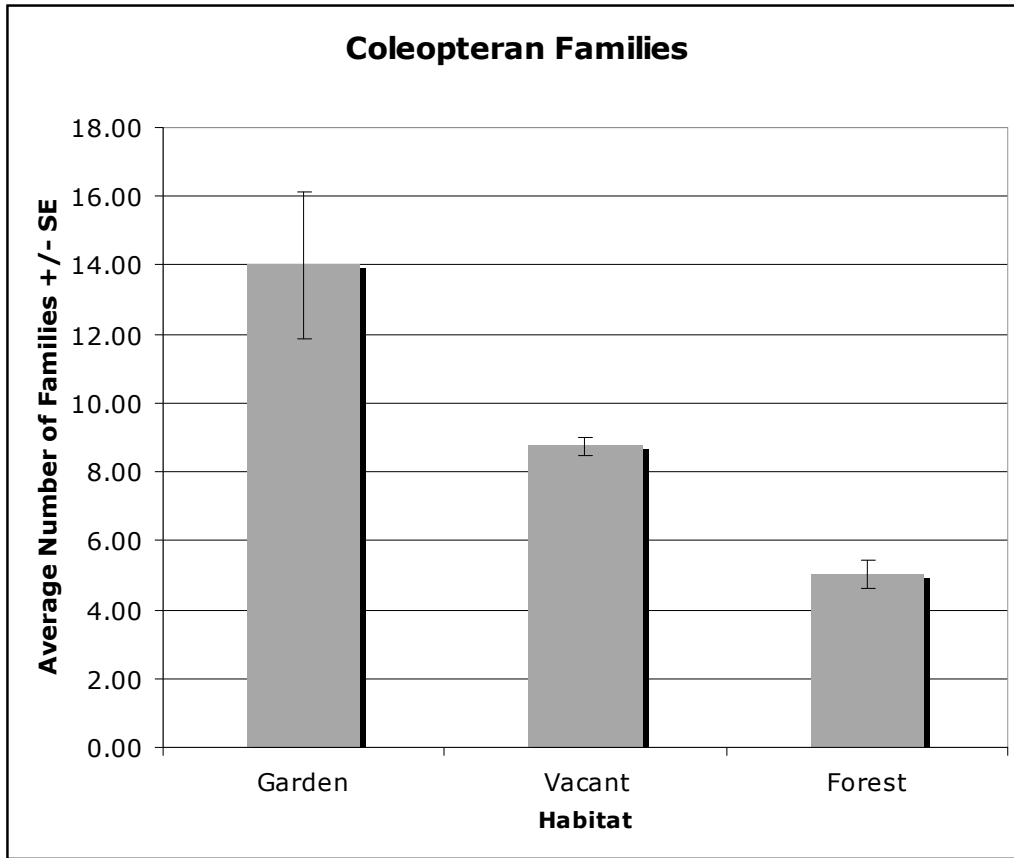


Figure 4. The average number of beetle families by habitat

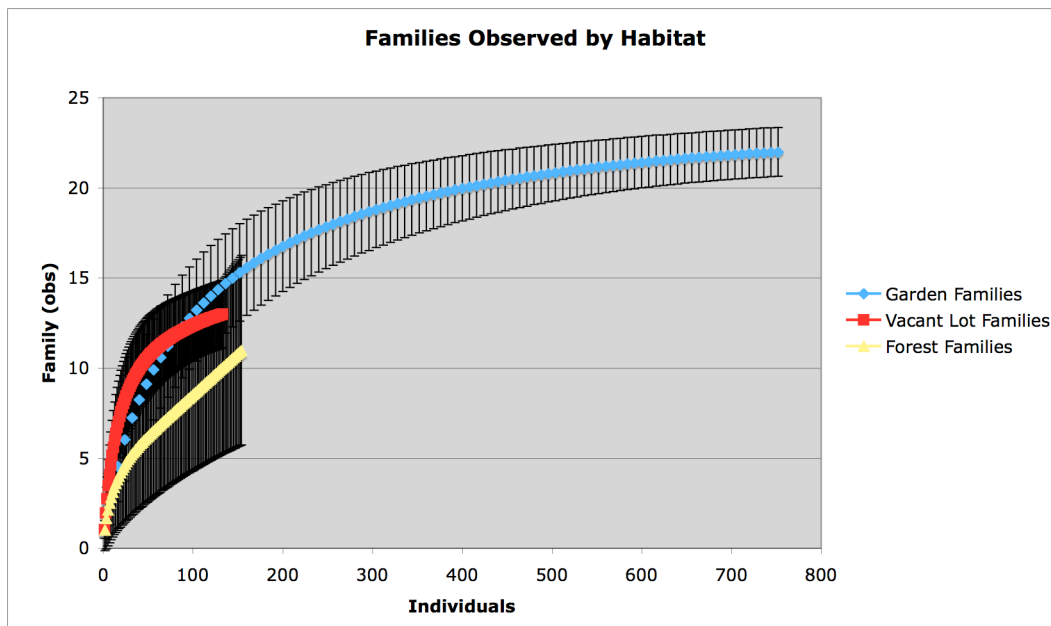


Figure 5. The species accumulation curves for beetle families captured in the study

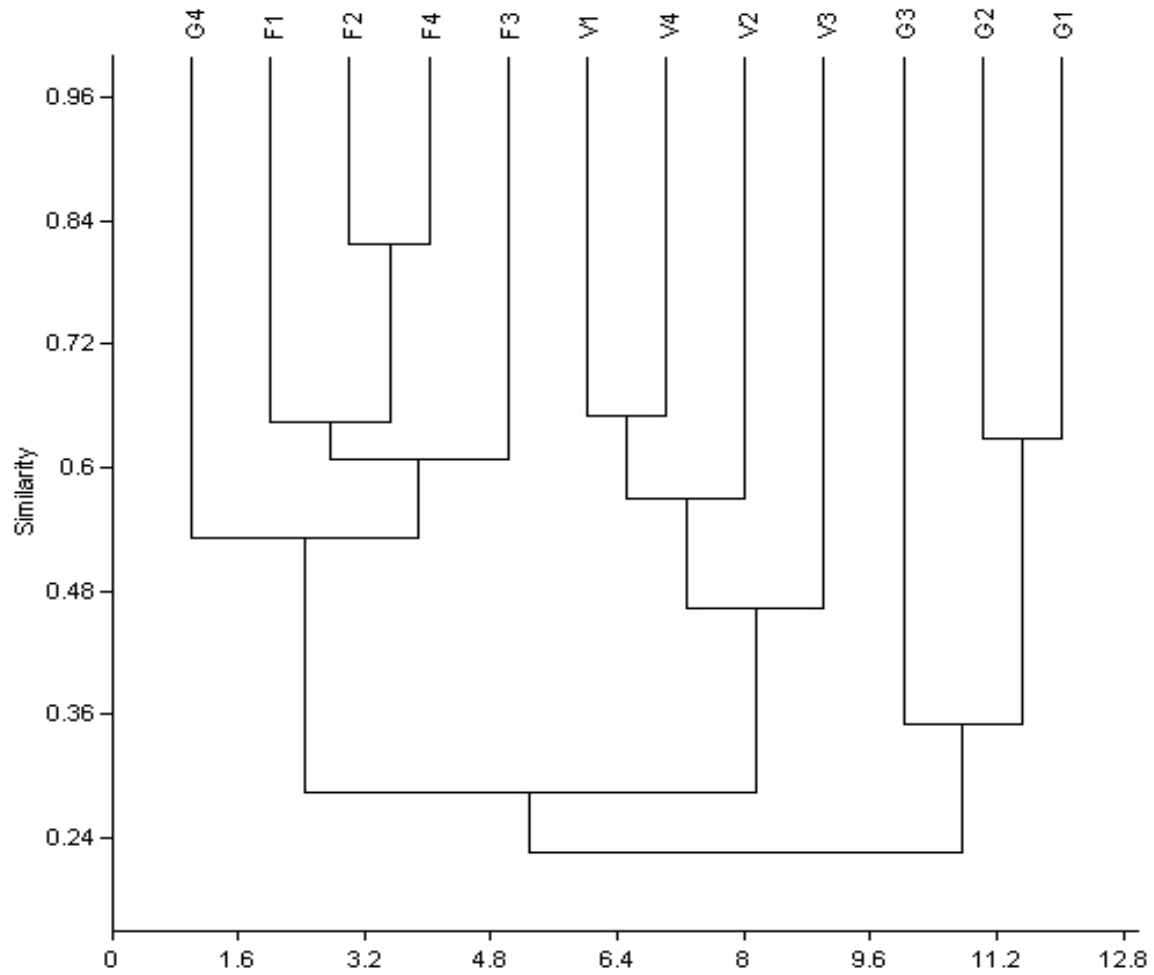


Figure 6. Bray-Cutis cluster analysis dendrogram for beetle families

| ANOSIM comparison | R-value | p-value |
|--------------------------|----------------|----------------|
| G vs. F | 0.7500 | 0.0288 |
| G vs. V | 0.8125 | 0.0283 |
| F vs. V | 0.9792 | 0.0264 |

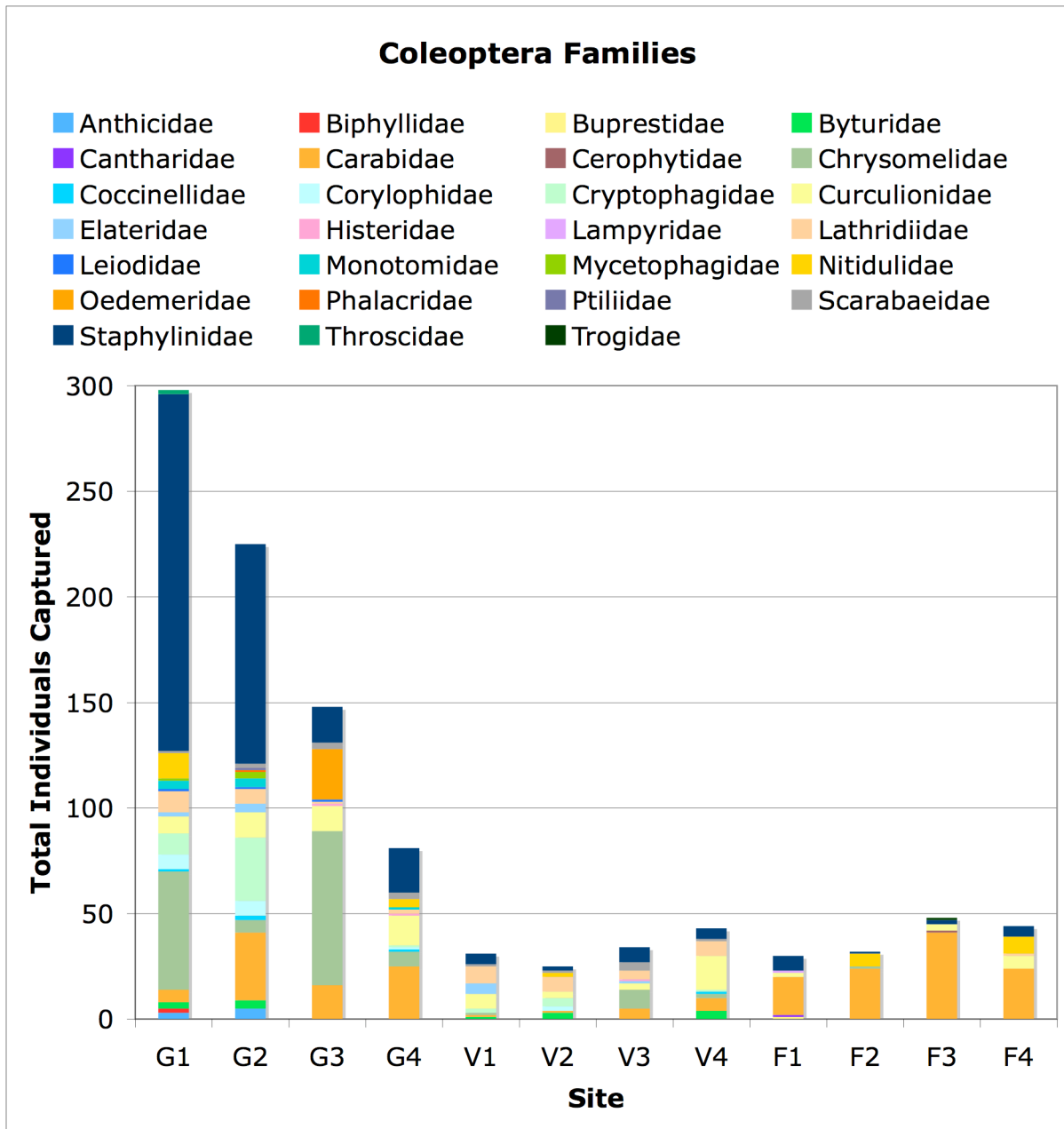


Figure 7. The number of individuals in each family at each site

| | | | | | | | | | | | | |
|-------|------|------|------|------|------|------|------|------|------|------|------|------|
| Total | G1 = | G2 = | G3 = | G4 = | V1 = | V2 = | V3 = | V4 = | F1 = | F2 = | F3 = | F4 = |
| | 298 | 225 | 148 | 81 | 31 | 25 | 34 | 43 | 30 | 32 | 48 | 44 |

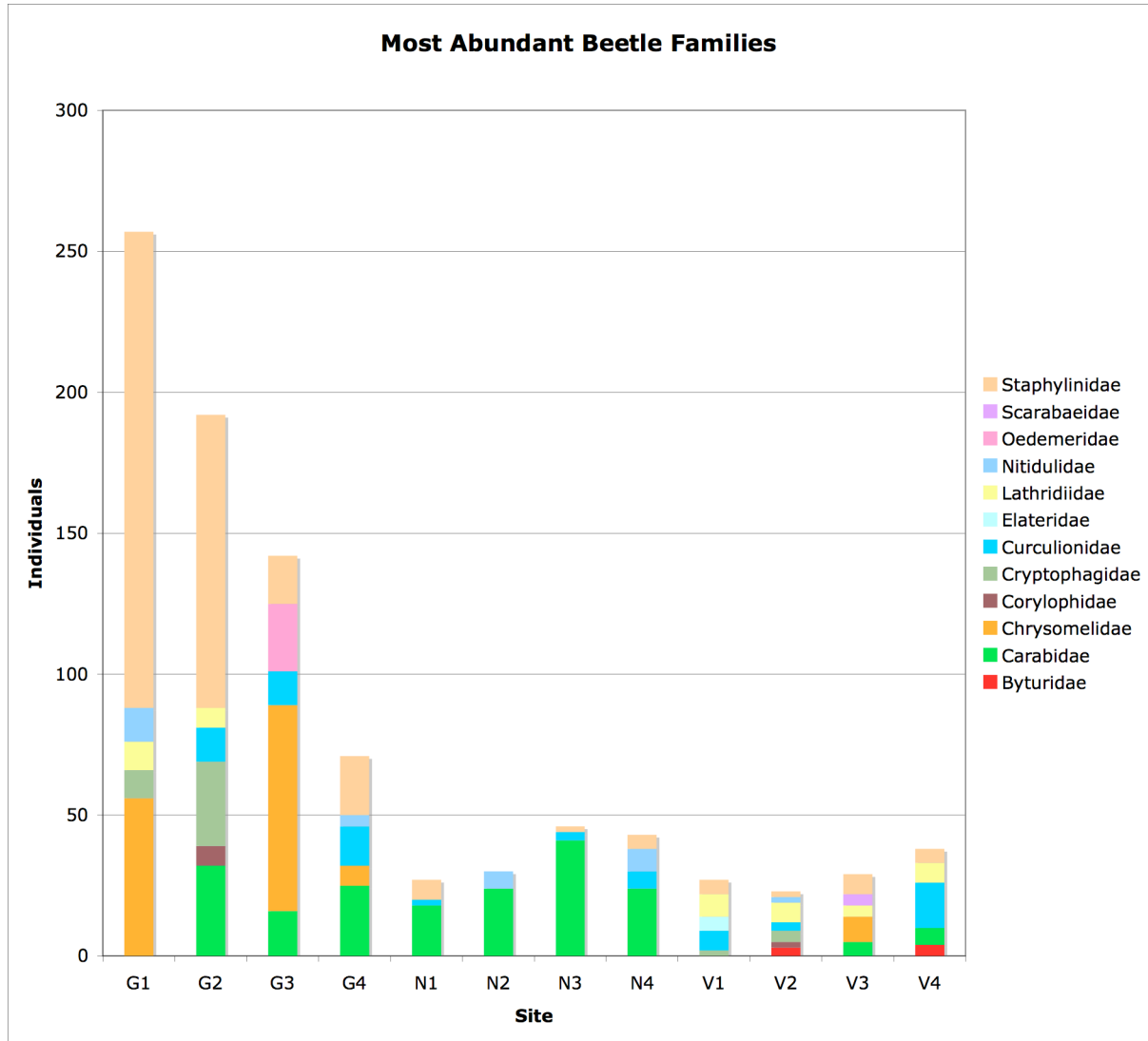


Figure 8. The five most abundant beetle families by site excluding families represented by a single individual

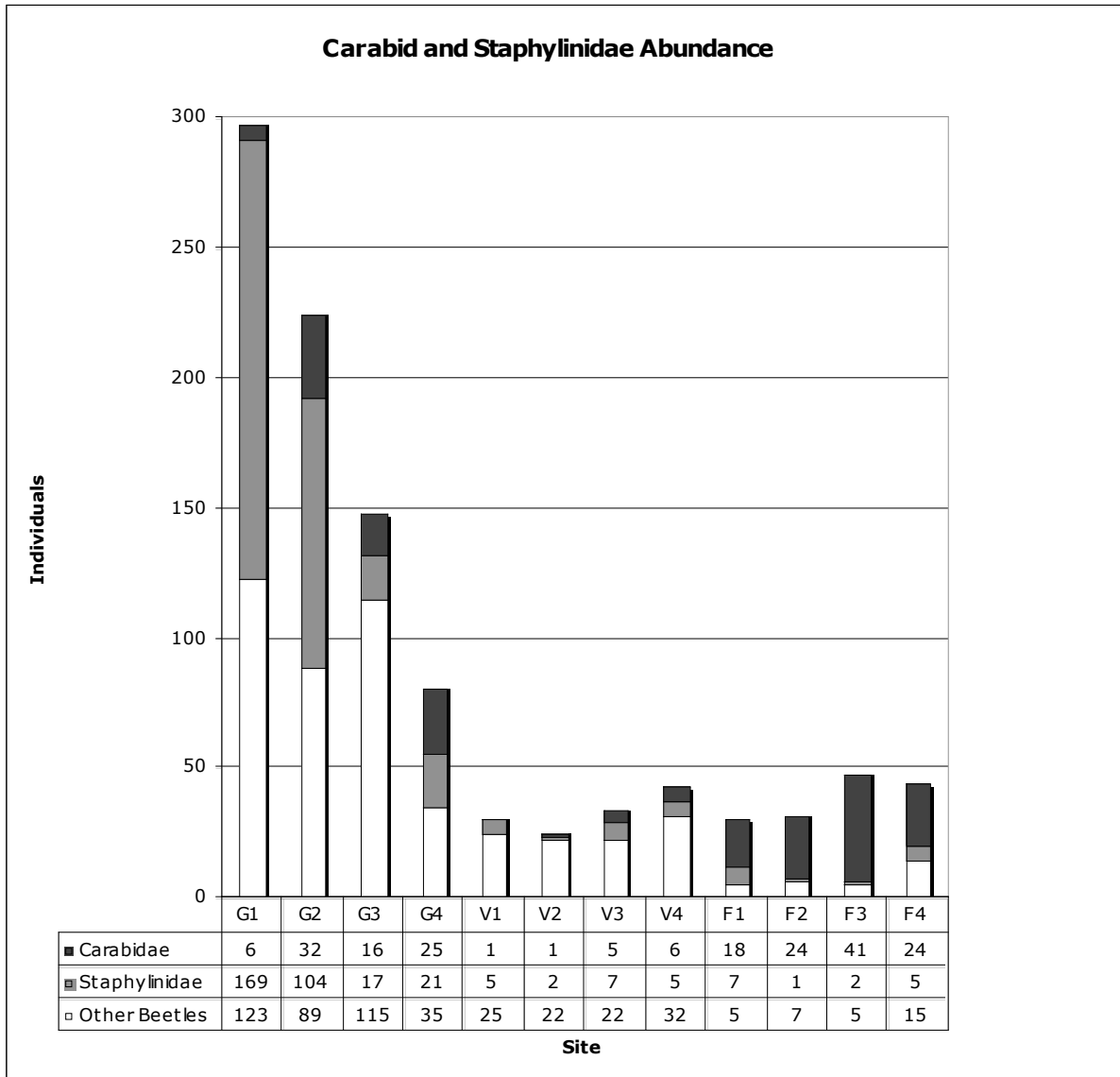


Figure 9. Carabid and rove beetle and rove beetle portion of abundance

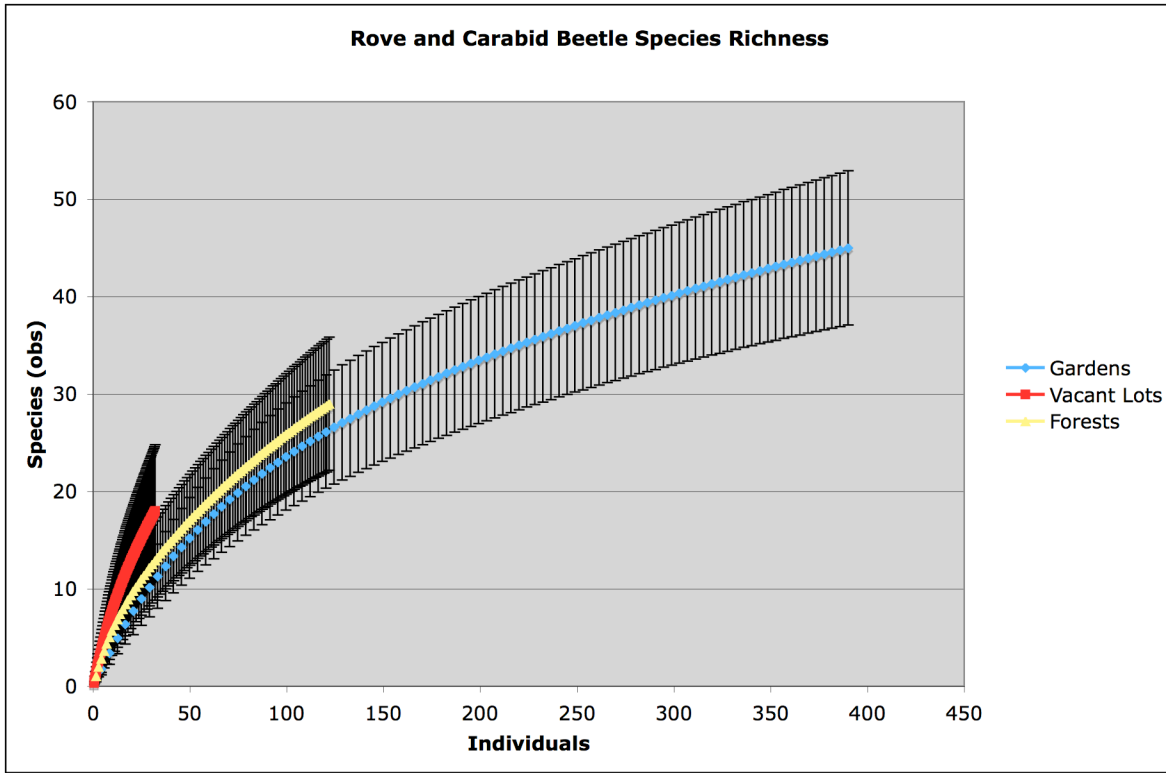


Figure 10. Species accumulation curve for the three habitat types based on the carabid and rove beetle capture

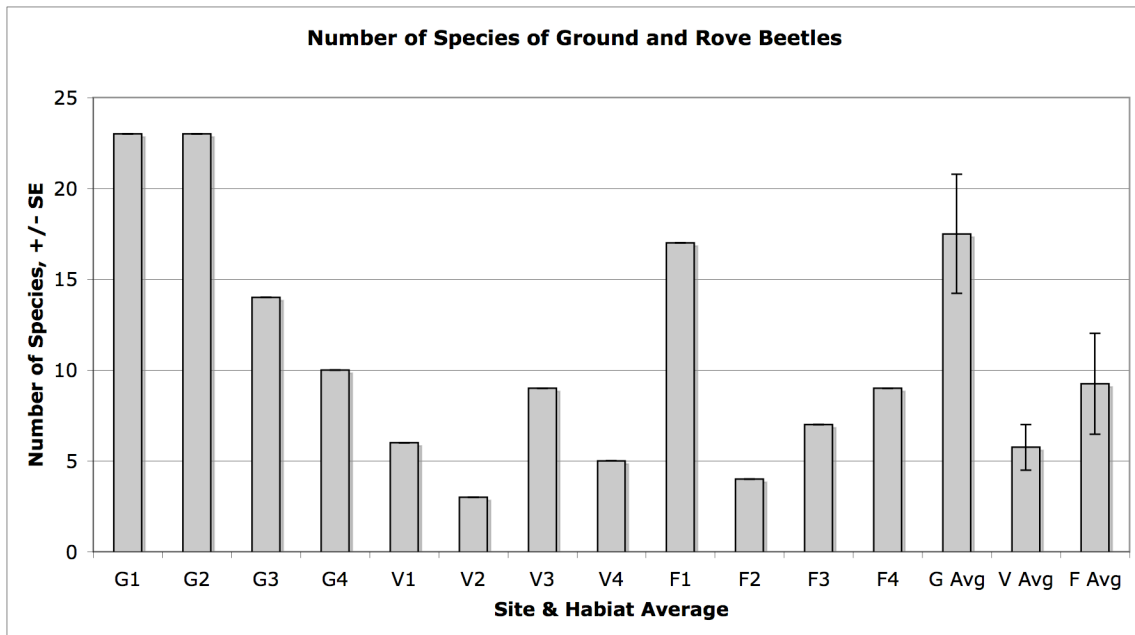


Figure 11. The observed number of carabid and rove beetle species by site and habitat

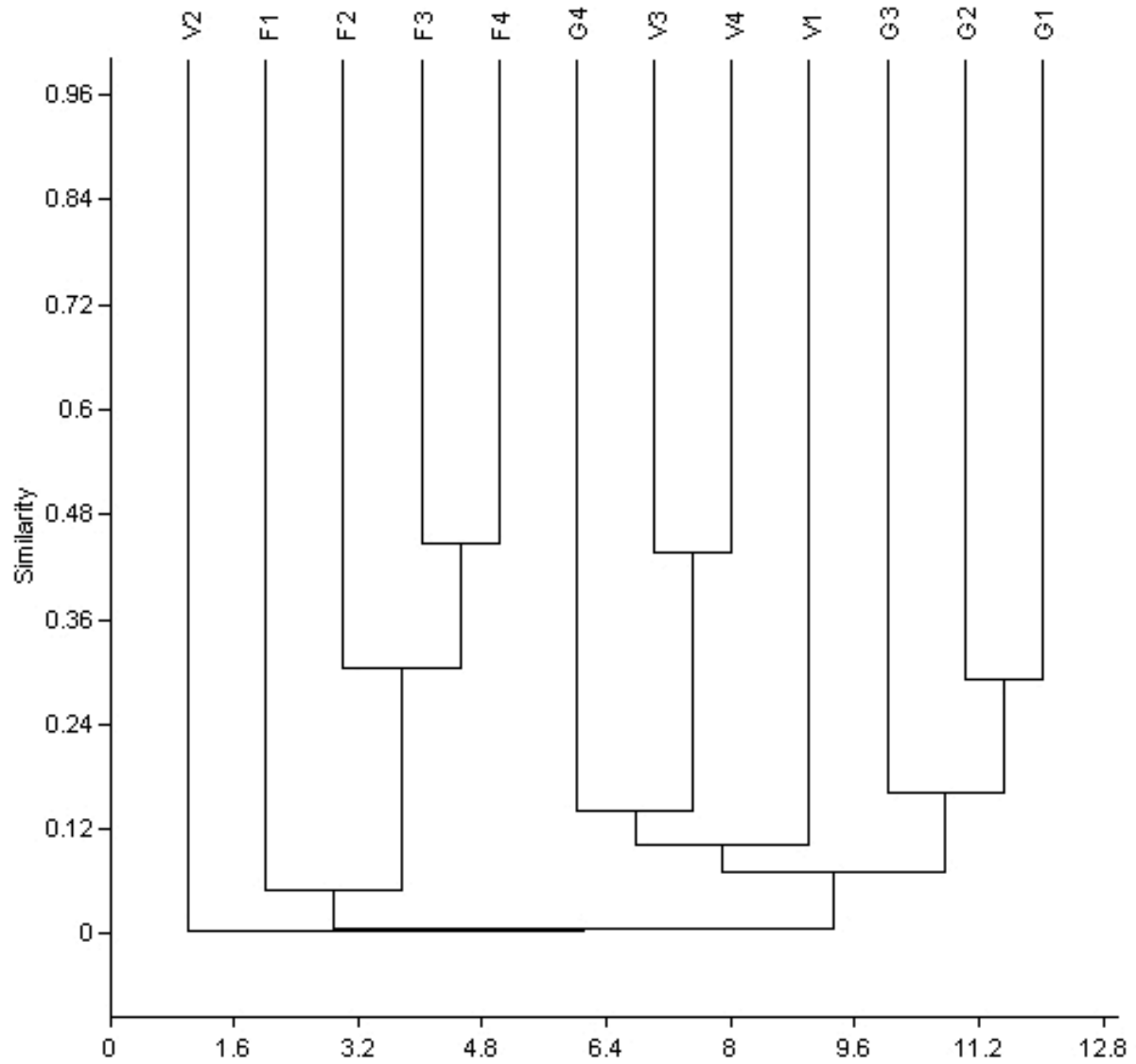


Figure 12. Bray-Cutis cluster analysis dendrogram for rove and carabid beetles

| ANOSIM comparison | R-value | p-value |
|--------------------------|----------------|----------------|
| G vs. F | 0.9792 | 0.0283 |
| G vs. V | 0.2500 | 0.0870 |
| F vs. V | 0.7135 | 0.0264 |

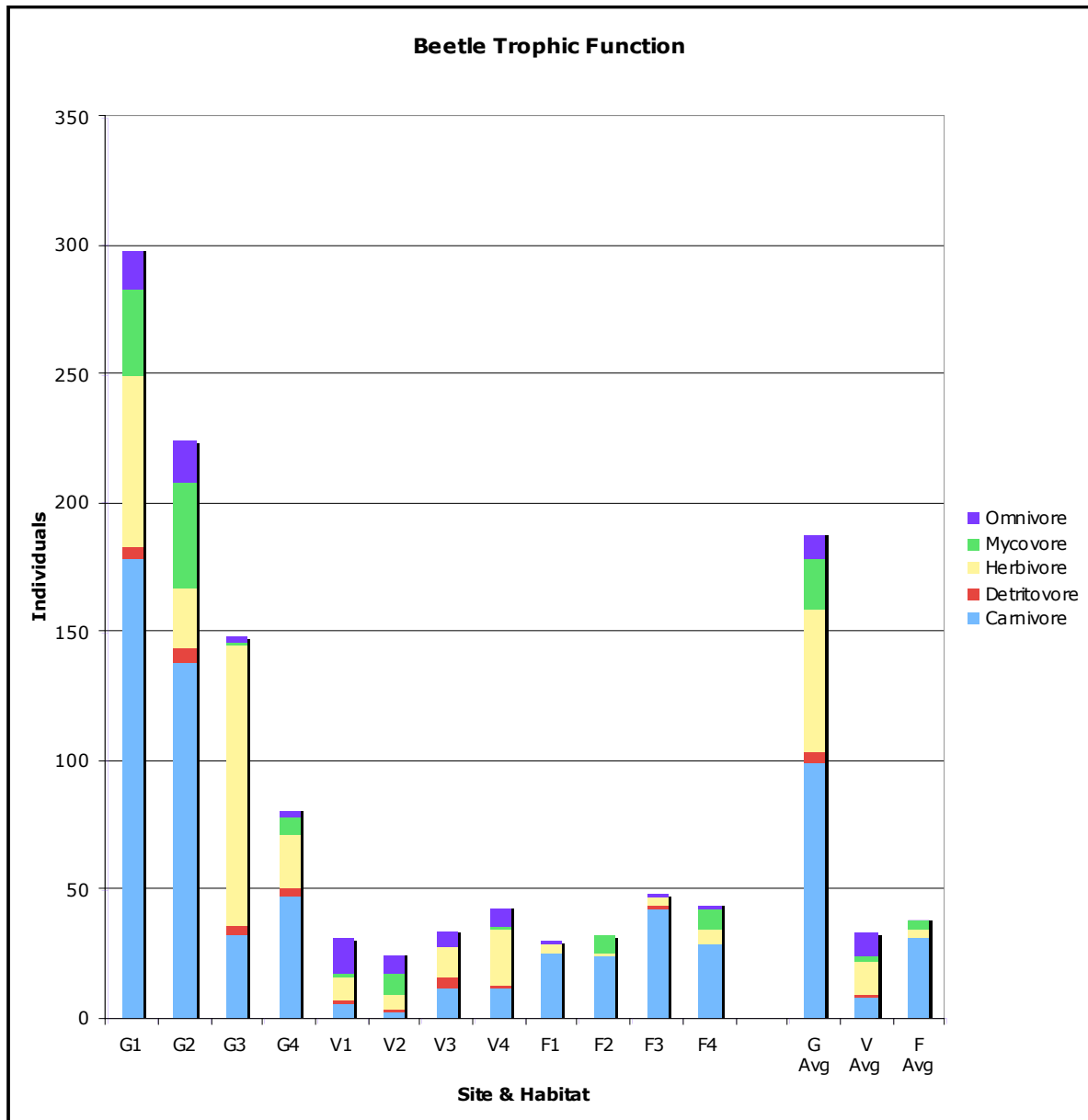


Figure 13. The number of individuals in each feeding guild by site and averaged for the habitats

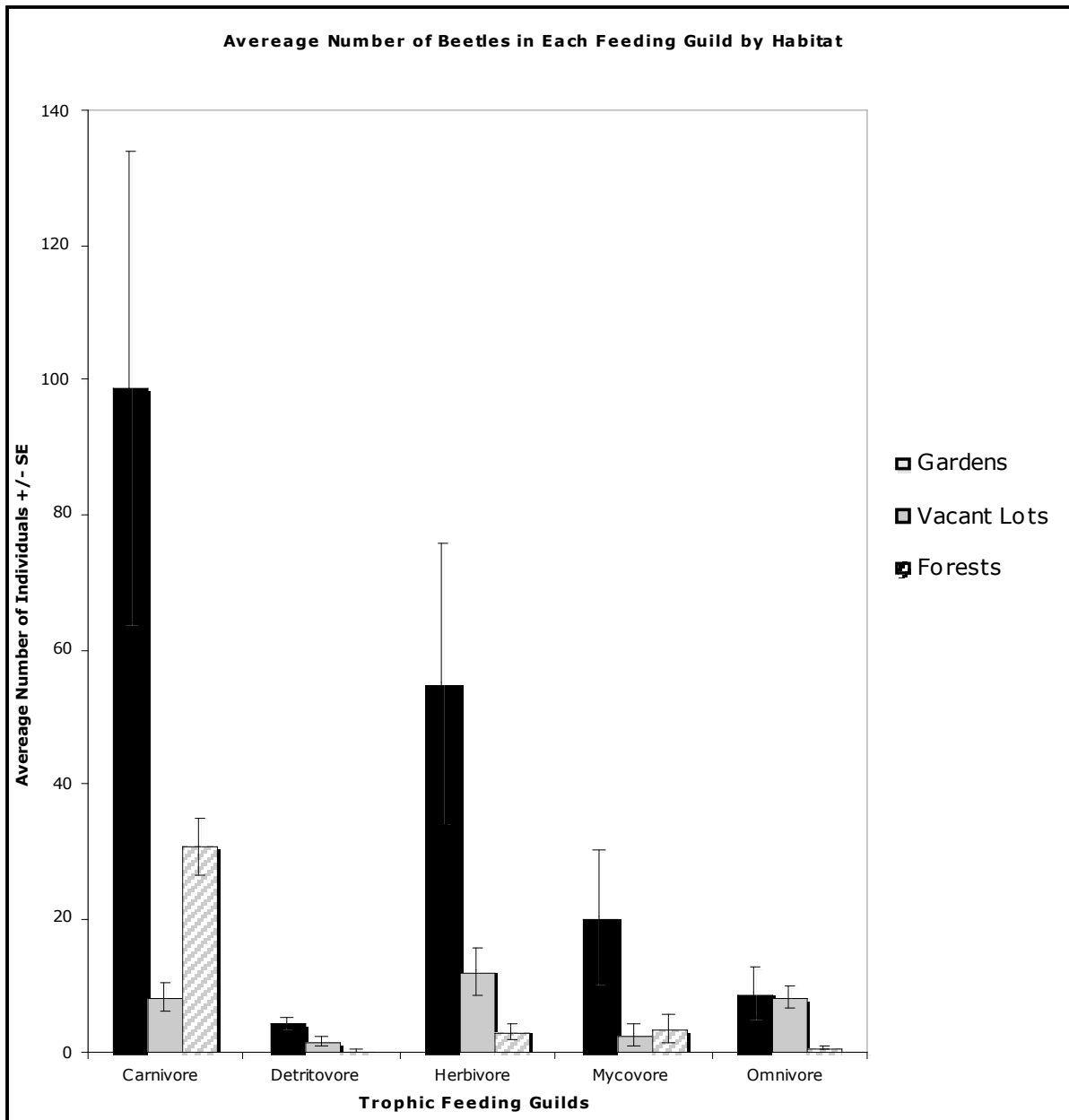


Figure 14. The number of individuals in each feeding guild by habitat

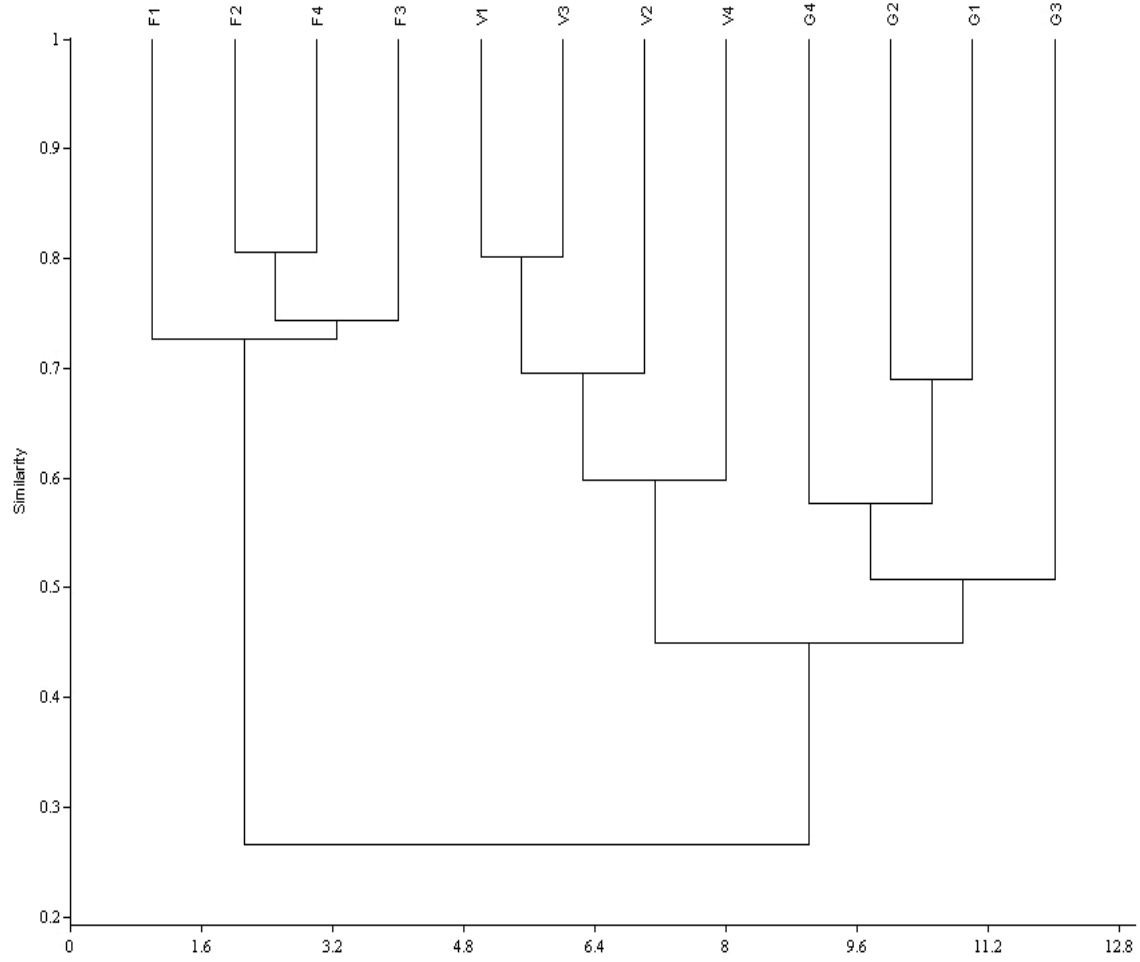


Figure 15. Bray-Curtis cluster analysis dendrogram of habitat factors, using UPGMA

Table 6. Tukey's Post-hoc Comparisons for Habitat Characteristics

| Habitat feature | Scale | Gardens | Vacant lots | Forests |
|---------------------------------|--------------------|---------|-------------|---------|
| % cover impermeable | 100 m ² | a* | a | b |
| % cover ground-level vegetation | 100 m ² | a, b | a | b |
| % cover shrub | 100 m ² | a | a | b |
| % cover bare ground | 100 m ² | a | a | a |
| % cover building base | 100 m ² | a | b | b |
| % average canopy cover | 100 m ² | a | a | b |
| number of trees | 100 m ² | a | a | b |
| number of buildings | 100 m ² | a | b | a |
| abundance of woody species | 400 m ² | a | a | b |
| richness of woody species | 400 m ² | a | a | b |
| density of trees | 400 m ² | a | a | b |
| average tree height | 400 m ² | a | a, b | b |
| average tree circumference | 400 m ² | a | a | a |
| shrub density | 400 m ² | a | a | b |
| average shrub height | 400 m ² | a | a | a |
| average shrub circumference | 400 m ² | a | a | a |
| average height of tallest plant | 1 m ² | a | a | a |
| % cover grass | 1 m ² | a | b | a |
| % cover bare ground | 1 m ² | a | a | a |
| % cover forbs | 1 m ² | a | a | a |
| % cover rocks | 1 m ² | a | a | a |
| % cover leaf litter | 1 m ² | a | b | b |
| % cover mulch | 1 m ² | a | b | b |
| % cover downed wood | 1 m ² | a | a | b |

*Identical letters represent no significant difference; different letters represent significant differences; habitats with two letters were not significantly different from the other habitats

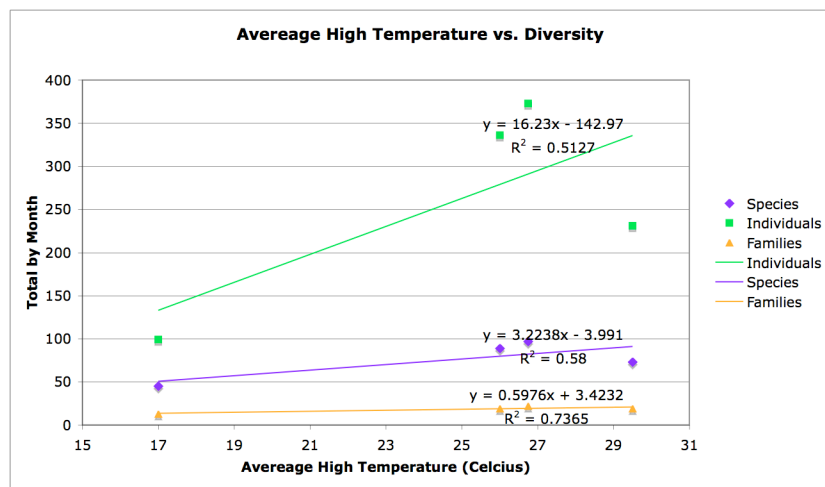
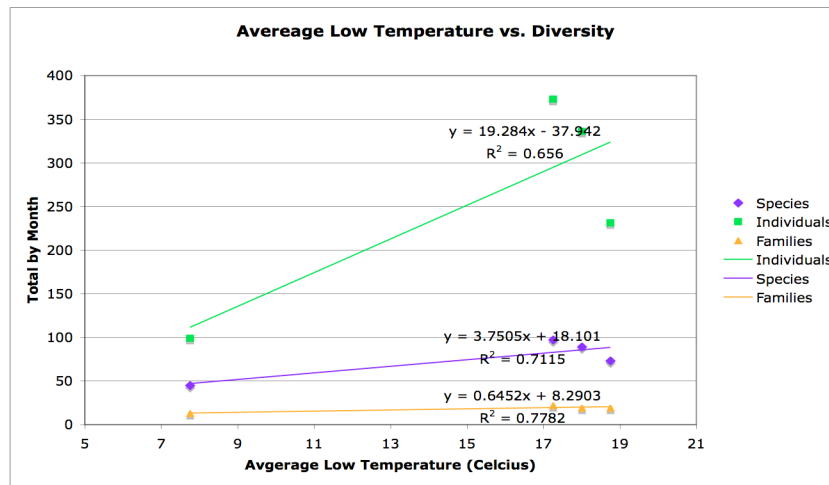
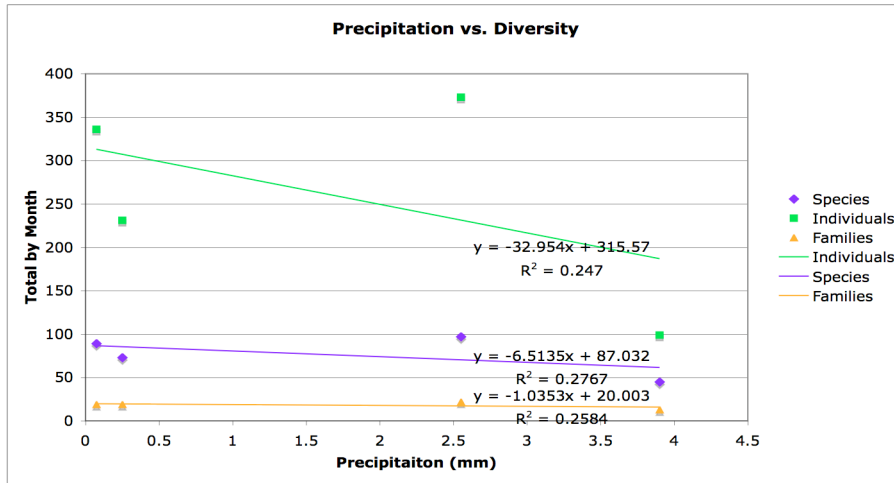


Figure 16 a. Total Precipitation vs. diversity measures

Figure 16 b. Average low temperature vs. diversity measures

Figure 16 c. Average high temperature vs. diversity measures

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CHAPTER IV

CONCLUSION

In this thesis, I examined the seasonal activity, species richness, habitat distribution and characteristics of beetles in the urban gardens, vacantly lots and remnant forests of Detroit. By dividing the beetles into different taxonomic groupings, I presented results that represent both underlying habitat differences on beetle assemblages as well as biases of the sampling and analytical methodologies.

Chapter II explored the seasonal activity, species richness, habitat distribution and characteristics of carabid beetles in the three habitats: gardens, vacant lots and remnant forests. Collecting this data was important for consideration of the effects of each of these habitats in maintaining the ecological integrity of the carabid assemblage. Carabid beetle activity did vary with season, with the greatest capture in August. The forests had a greater abundance of native species with primarily predatory feeding habits, and more of these beetle species were flightless when compared to other habitats, indicating more that the forests are more stable habitats than the gardens or vacant lots. T species richness curves suggest that additional sampling would need to be employed to have a more accurate count of species of carabid beetles in all three habitats. The capture of carabids by pitfall trapping in the vacant lots was very low, suggesting that it is a poor habitat for carabid beetles. However, other sampling methodologies should be employed to confirm this.

In general, the carabid genera of the forest habitats were not found in either the gardens or the vacant lots. The forest of Belle Isle was of particular distinction, as it housed the most species of carabid beetles, all of which were native to North America, as well as one individual that represents a new state record, *Pterostichus praetermissus*. The remaining forests hosted a smaller diversity of carabids that included several exotic species, such as *Carabus nemoralis* found in both River Rouge and Palmer parks, and *P. melanarius* in Balduck and River Rouge parks, and whose dimorphic trait might be of interest when considering the stability of a habitat.

A few other species of carabids merit mention. *Cyclotrachelus sodalis* (*sodalis*) was the only species found in both a forest (River Rouge) and a vacant lot habitat, but only one individual was found in vacant lot. As these two sites were near each other in proximity and this

beetle does not fly, this species might be is a possible candidate for a ground migration study; perhaps this beetle in moving from one high quality habitat to another through the vacant lots, or has a range that spans more than one habitat type. *Scartius subterraneus* appears to be a gregarious carabid in gardens, and may be considered as a primarily beneficial inhabitant as it is primarily carnivorous (though it may occasionally eat seedlings).

Chapter III considered three alternate taxonomic groupings within Coleoptera - all morphospecies, families, and a subset including only the carabid and rove beetles - in order to better evaluate the overall abundance and distribution of species in the urban setting, and to consider how these alternate groupings of the beetles change the resulting assessment of the site similarity. I also examine the feeding guild associations of the families and climactic variables in order to better understand the observed differences in seasonal activity and family abundance. The activity level of beetles peaked in July, likely when garden resources are at their height, and then slightly declined in August – this change did not appear to be coupled with temperature after the month of June, perhaps due to the accumulation of degree-days for hatching or the appearance of attractive food resources. The overall beetle activity-density in forests and vacant lots was very similar. The activity-density and species richness of beetles in gardens far exceeded that of the other habitats, with predatory and herbaceous beetles being significantly more numerous in these settings. More families are shared between vacant lot habitats and the gardens than the forests and gardens or the forests and the vacant lots. The forests had more families that were exclusive to this habitat, and were represented by a single individual, indicating a great number of beetles in this setting are yet to be encountered.

Based on vegetative and land cover habitat characteristics alone, the sites grouped into habitat clusters without any misplaced branches on the dendrogram representation. However, when grouped by the activity of beetles based on the three different taxonomic schemes, three different groupings emerged. Acres of Hope Garden (G4) varied in its position in cluster analysis. If the beetle families were analyzed, the garden is more similar to the forests due to the prominence of carabid beetles in this garden. When the carabid and rove beetles or all the morphospecies are analyzed, G4 clustered with the vacant lots - likely because it shared rove beetle species with vacant areas. Acres of Hope Garden a small garden set near several vacant lots, and is relatively distant from other garden sites. The lack of shared species with other

gardens could possibly be due the fact that garden is too small to maintain food for carnivorous carabids, and is therefore inhabited by carabids that migrate from surrounding vacant lot areas.

Though this study represents only a snapshot of the life that has endured or prospered in Detroit greenspaces after 300 years of human use, it lays a foundation for mythology that might be utilized in other sites, or even as part of a greater effort to discover the abundance and distribution of beetles in the city. Ongoing efforts in Canada and in association with the Central Arizona-Phoenix Long Term Ecological Research project (Nature Canada 2007; Global Institute of Sustainability and Arizona State University 2008), are introducing this concept to students, and offering the public the opportunity to become involved in the effort. In association with the Audubon society, citizen-scientist have successfully gathered the most abundant and exact data in most areas for birds in the region (Colon and Stouffer 2009). Other ongoing species assessment projects, such as those at Humbug Marsh, a coastal marsh on the Michigan mainland of the Detroit River part of the Detroit River International Wildlife Refuge, could contribute to such an effort for insects in Detroit (Manny 2008). Andow suggests that the evolutionary theory of related taxa may be the most valuable way to consider the relationship between arthropods and vegetation diversity (1991). Mapping the distribution of taxa may help to separate the internal, habitat quality factors from the surrounding effects of the matrix (Rickman and Connor 2003), or may allow for the identification of indicator species that could be used to detect environmental changes such as the effects of pollution.

Efforts to reforest the city through tree planting projects and environmental education (The Greening of Detroit 2004-2008) are a valuable contribution to Detroit. This study and other studies in San Francisco suggests that lessening the extent of urbanization by increasing the tree canopy cover may assist in increasing the abundance of certain species of beetles, such as carabid beetles, which are more abundant in forested areas (Connor et al. 2003). Several SE Michigan institutions, including Southeast Michigan Council On Governance (SEMCOG), City of Detroit, and the Greening of Detroit have begun to plan and construct greenways that link natural attractions and popular destinations. Such forest restoration efforts, if funding is available, may also be a way to help convert “blighted” areas to a more valuable use. However, few biological indices exist for terrestrial resources, and restoration efforts are often hampered by their lack of evaluative methods; a biological index may assist in defining “what to restore to”

after intense land degradation (Bratton 1992), and in adaptive management and evaluation of terrestrial restoration (Karr and Kimberling 2003). This research could be applied towards applications of biological work that explore creating a terrestrial “Index of Biological Integrity” for Detroit (Karr and Kimberling 2003; Kimberling and Karr 2001, 1999). Alternatively, urban gardeners may benefit from a biological index that allows them to evaluate the health of their garden area to maximize ecosystem services such as biocontrol by predatory beetles.

Finally, the great diversity of beetles found in urban gardens suggests that they are a significant contributor to the regional biodiversity of the Detroit urban area. Though the studies do not suggest that gardens serve as habitat for beetles migrating between natural areas such as the forests, the number of beetles harbored in gardens may be a source for diversity in the sea of relatively low-quality ruderal vacant lot vegetation. Several gardens, such as Earthworks urban farm, already incorporate the existing diversity into garden educational programs for children. With the additional benefits of fostering healthier diets and entrepreneurial experiences, creating gardens and related scientific and environmental education programming in urban Detroit schools appears to be one feasible approach to improving an ailing school system (Lieberman and Hoody 1998).

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