Table of Contents

<table>
<thead>
<tr>
<th>Title</th>
<th>Author</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Patterns of spatial distribution of Japanese Barberry in the Big Woods</td>
<td>Rachel Cable</td>
<td>5</td>
</tr>
<tr>
<td>The impact of habitat transitions and edge effects on hickory-</td>
<td>Susan Cheng</td>
<td>12</td>
</tr>
<tr>
<td>feeding leaf miners</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Survival of the fittest: competitive tree community dynamics in forest</td>
<td>Sahar Haghighat</td>
<td>20</td>
</tr>
<tr>
<td>undergoing successional change</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nesting site limitations and nesting resources of cavity nesting</td>
<td>Cindy Bick</td>
<td>26</td>
</tr>
<tr>
<td>ants (Hymenoptera: Formicidae) in the E.S. George Reserve, MI</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Source-sink dynamics of <em>Peromyscus leucopus</em> population in the George</td>
<td>Ana Chará</td>
<td>34</td>
</tr>
<tr>
<td>Reserve</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Influence of canopy diversity on decomposition of leaf litter in a</td>
<td>Rafael D’Andrea</td>
<td>39</td>
</tr>
<tr>
<td>temperate forest</td>
<td></td>
<td></td>
</tr>
<tr>
<td>This is ants not art: determining competition and dominance in ant</td>
<td>Lauren Maloney</td>
<td>45</td>
</tr>
<tr>
<td>species using spatial modeling</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Variance of fungal diversity and abundance along plant community,</td>
<td>Iman Sylvain</td>
<td>51</td>
</tr>
<tr>
<td>substrate, and functional lines</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Application of ant spatial dynamics to modeling scalar patterns of</td>
<td>William Webb</td>
<td>62</td>
</tr>
<tr>
<td>competition</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Acarodomatia of woody plants of the E.S. George Reserve</td>
<td>David Gonthier</td>
<td>71</td>
</tr>
<tr>
<td>Spatial distribution of barberry in the E.S. George Reserve</td>
<td>John Guittar</td>
<td>80</td>
</tr>
<tr>
<td>Liana diversity and abundance following succession in the E.S.</td>
<td>Abigail Hyduke</td>
<td>86</td>
</tr>
<tr>
<td>George Reserve</td>
<td></td>
<td></td>
</tr>
<tr>
<td>An investigation of flight initiation response in various animals at</td>
<td>Mingqi Liu</td>
<td>94</td>
</tr>
<tr>
<td>the E.S. George Reserve</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Health and biodiversity of the Honey Creek watershed</td>
<td>Kassandra Semrau</td>
<td>100</td>
</tr>
<tr>
<td>Bone assemblages in the E.S. George Reserve</td>
<td>Kate Zemenick</td>
<td>107</td>
</tr>
<tr>
<td>Black/red oak regeneration strategies: do seedlings</td>
<td>Rafael D’Andrea</td>
<td>115</td>
</tr>
<tr>
<td>intelligently allocate biomass in response to stress?</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Title</td>
<td>Author</td>
<td>Page</td>
</tr>
<tr>
<td>----------------------------------------------------------------------</td>
<td>-------------------------</td>
<td>------</td>
</tr>
<tr>
<td>Leaf trichomes, acarodomatia, and mite abundance on temperate woody</td>
<td>David Gonthier</td>
<td>121</td>
</tr>
<tr>
<td>plants</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Inquilene abundance in relation to external variables of <em>Sarracenia</em></td>
<td>Abigail Hyduke</td>
<td>129</td>
</tr>
<tr>
<td>purpurea of hidden lake bog at the E.S. George Reserve</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Comparison of terrestrial snail diversity in mixed forest and</td>
<td>Binbin Li</td>
<td>137</td>
</tr>
<tr>
<td>manipulated red pine (<em>Pinus rubra</em>) forest</td>
<td></td>
<td></td>
</tr>
<tr>
<td>The effect of nutrient addition on species abundance and</td>
<td>Iman Sylvain</td>
<td>148</td>
</tr>
<tr>
<td>diversity in two experimental ponds at the E.S. George Reserve</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Differential response to microhabitat variability in seedling</td>
<td>William Webb</td>
<td>156</td>
</tr>
<tr>
<td>growth strategies of early and late successional species</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tree and leaf color ratios in the Big Woods, E.S. George Reserve</td>
<td>Kate Zemenick</td>
<td>162</td>
</tr>
<tr>
<td>An analysis of Big Woods tree demography using empirical data and a</td>
<td>Rafael D’Andrea</td>
<td>169</td>
</tr>
<tr>
<td>size-structured population model</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fishy cuisine: comparing overlap in the diets of the tubenose</td>
<td>Sahar Haghighat</td>
<td>175</td>
</tr>
<tr>
<td>goby and the round goby with various other native benthic species in</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lake Erie</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Survey of the ants of Bee Island: 60 years of change</td>
<td>Kassandra Semrau</td>
<td>182</td>
</tr>
<tr>
<td>A re-examination of <em>Peromyscus leucopus</em> demography and source-sink</td>
<td>Cindy Bick</td>
<td>190</td>
</tr>
<tr>
<td>dynamics on the E.S. George Reserve</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Owls surveys in the E.S. George Reserve</td>
<td>Rachel Cable</td>
<td>196</td>
</tr>
<tr>
<td>Effect of barberry on leaf litter associated invertebrates</td>
<td>Ana Chará</td>
<td>202</td>
</tr>
<tr>
<td>The effect of disturbance recovery time on forest leaf litter</td>
<td>Susan Cheng</td>
<td>208</td>
</tr>
<tr>
<td>Obtaining and applying simple estimates of sapling biomass</td>
<td>Rafael D’Andrea</td>
<td>213</td>
</tr>
<tr>
<td>Pond connectivity and the abundance of adult salamanders in</td>
<td>David Gonthier</td>
<td>221</td>
</tr>
<tr>
<td>the E.S. George Reserve</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Berberis thunbergii</em>: the past, present, and future spatial</td>
<td>John Guittar</td>
<td>229</td>
</tr>
<tr>
<td>distribution of an invasion</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Title</td>
<td>Author</td>
<td>Page</td>
</tr>
<tr>
<td>----------------------------------------------------------------------</td>
<td>----------------------</td>
<td>------</td>
</tr>
<tr>
<td>Herbaceous understory diversity as a function of uppercanopy cover and as a determinate of tree competitive community dynamics</td>
<td>Sahar Haghighat</td>
<td>237</td>
</tr>
<tr>
<td>Do pollinators prefer late-bloomers? Late blooming flowers of the E.S. George Reserve and their abilities to attract pollinators</td>
<td>Abigail Hyduke</td>
<td>242</td>
</tr>
<tr>
<td>An investigation of autumn maple leaf color in the E.S. George Reserve</td>
<td>Mingqi Liu</td>
<td>249</td>
</tr>
<tr>
<td>A prediction of potential disperses for adhesive seeds</td>
<td>Binbin Li</td>
<td>260</td>
</tr>
<tr>
<td>Does temperature kill the intransitive loop: the effect of temperature on dominance of ant species, <em>Tetramorium caespitum</em>, <em>Monomorium minimum</em>, and <em>Solenopsis molesta</em></td>
<td>Lauren Maloney</td>
<td>268</td>
</tr>
<tr>
<td><em>Berberis thunbergii</em> and <em>Elaeagnus umbellata</em> in the field and forest</td>
<td>Kassandra Semrau</td>
<td>273</td>
</tr>
<tr>
<td>Liana relay ascension at the E.S. George Reserve</td>
<td>Iman Sylvain</td>
<td>279</td>
</tr>
<tr>
<td>Commensal properties of native red maple seedlings and an invasive shrub, Japanese Barberry, in the forest understory</td>
<td>William Webb</td>
<td>288</td>
</tr>
<tr>
<td>Phenotypic variation of <em>Prunus serotina</em> bark: implications for the insect community</td>
<td>Kate Zemenick</td>
<td>295</td>
</tr>
</tbody>
</table>
PATTERNS OF SPATIAL DISTRIBUTION OF JAPANESE BARBERRY IN THE BIG WOODS

RACHEL CABLE, RAFAEL D’ANDREA, KC SEMRAU, IMAN SYLVAIN, JOHN VANDERMEER & KATE ZEMENICK

ABSTRACT

Japanese barberry, *Berberis thunbergii*, is an invasive shrub species that is known as a dominant understory species in the Big Woods plot of the E. S. George Reserve. However, the distribution and clumping within the plot is not known. To acquire this distribution, a sample area of the Big Woods was divided into quadrats, which were each surveyed over the period of one day. The percent ground coverage of each quadrant by *B. thunbergii* was estimated and categorized, and a final visual representation of the entire sample area was made of the categorizations of all quadrats. The visual representations show clear patterns of distribution of *B. thunbergii* across the Big Woods plot, including higher overall densities north of Esker Rd.

INTRODUCTION

Much of the documentation on the Big Woods plot in the George Reserve focuses on the distribution of canopy and sub-canopy species including red maple, black cherry, witch-hazel and a variety of hickory and oak species. In recent surveys of the Big Woods plot, it has become clear that the forested community composition is changing. Notably, we have entered a period of turnover in the dominant canopy tree species. Despite the wealth of information on the community composition and distribution of canopy tree species in the Big Woods plot, little is known about the dynamics and distribution of the understory species, particularly the invasive shrub *Berberis thunbergii*.

*Berberis thunbergii* (Japanese barberry) is a temperate, deciduous shrub native to Japan. The berries of *B. thunbergii* are dispersed by birds and small mammals, and it has a high germination rate of nearly 90% (Swearingen, 2009). It is able to spread vegetatively through arching branches that establish roots after making contact with the ground, thus establishing areas of high clonal density. Additionally, *B. thunbergii* is capable of increasing nitrogen availability in the soil, creating a positive feedback that enables it to spread more successfully (Ehrenfeld et al, 2001). *B. thunbergii* has been cultivated for landscaping due to its attractive morphology and highly tolerant and adaptable nature. It was first brought to North America in 1875 and planted in the Arnold Arboretum in Boston (Swearingen, 2009). It was soon promoted as a replacement for *Berberis vulgaris* (common barberry) and has since spread quickly throughout the eastern United States and Canada. It has been observed that *B. thunbergii* is a dominant understory species in the Big Woods plot of the E.S. George Reserve (Grimm et al. 2009).

*Berberis thunbergii* is thought to behave as persistent pioneer species that spreads readily through open disturbed habitats such as abandoned agricultural fields (DeGasperis and Motzkin, 2007). Although *B. thunbergii* prefers open areas, it has a high tolerance for shade, which allows
it to persist in forested areas with high canopy cover (Silander and Klepeis, 1999), where it is found most on the George Reserve. Higher growth in forested areas may be due to competition with other invasive species in open areas, such as *Elaeagnus umbellata*, which is commonly found in old fields in the George Reserve.

Because of its tendency to spread clonally via branches touching the ground and rooting, *B. thunbergii* is predicted to exhibit patterns of clumping where plants grow within branch-length of each other. By conducting a survey of the relative percent ground cover of *B. thunbergii*, we hope to determine the extent to which *B. thunbergii* has infiltrated the Big Woods and at what scale (if any) it displays clumping patterns. We will observe whether plants grow closely to one another in clumps and if those clumps grow in close proximity to each other in certain areas. If *B. thunbergii* does grow in such thick and close clumps, it may prevent the establishment of tree seedlings by crowding resources such as light and nutrients. In addition to density, we will examine whether the distribution of *Berberis thunbergii* is correlated with particular canopy/sub-canopy species, water availability, or terrain. By taking these observations, we hope to see if *B. thunbergii* growth in the Big Woods seems to take over the entire forest floor, or if it is restricted by some environment factors.

**MATERIALS AND METHODS**

The Big Woods had been previously demarcated into hectare-sized plots with a set of stakes 100m apart, forming a 400m X 500m grid (Fig.1, Allen et al., 2009). A 400m X 400m section of these plots was further divided into sixty-four 50m X 50m quadrats. Upon visual inspection of the quadrat area, multiple observers estimated the percent ground coverage by *Berberis thunbergii*. The average of these estimates was then used to place the quadrat in one of five categories: 1 (0%-19% coverage), 2 (20%-39%), 3 (40%-59%), 4 (60%-79%), or 5 (80%-100%). Quadrats were progressively sampled by either viewing the entire quadrat from its perceived center or walking transects parallel to the north and south quadrat boundary lines.

Figure 1. Map of sampled area of the Big Woods (Allen et al 2009).
RESULTS

Results of our survey show that *Berberis thunbergii* is exhibiting a clumped distribution in the Big Woods plot of the George Reserve. Graphs were created in Excel to display the relative percent coverage of each 50m x 50m plot (Fig. 2). These representations were then compared to a topographical map of the sample area (Fig. 3) and data on the distribution of black cherry, red maple, and witch-hazel trees within the sample area.

Figure 2. Visual representations of percent ground coverage by *B. thunbergii* in the sampled area. (top) Percent coverage represented by relative size of circles. (bottom) Percent coverage represented by color intensity (yellow to red). Quadrat borders demarcated every 50 meters.
Figure 3. General distribution of the clumps of *B. thunbergii* in the Big Woods on a topographical map of the sampled area.

Figure 4. The distribution of Black Cherry in relation to the distribution of *B. thunbergii* (left).

Our measure of *B. thunbergii*'s percent coverage shows that *B. thunbergii* is present in the majority of the quadrats in the Big Woods plot (Fig. 2), with the highest percentage are found on the north side of Esker Rd and south of the “highlands” of the northern end of the plot. Our results indicate that *B. thunbergii* does have a clumped distribution in the Big Woods plot. In
comparison to the topographic landscape of the plot, higher ground coverage occurs in low areas, on steep hills along Esker Rd., and in marshland (Fig. 3).

In comparison with the plots of red maple and witch-hazel found within the sample area, we found no positive or negative correlation with our plot of *B. thunbergii*. However, the plot of black cherry did show a negative correlation with our plot of *B. thunbergii* (Fig. 4).

**DISCUSSION**

After our survey of the Big Woods plot, we have found that *B. thunbergii* seems to be clumped at a scale much greater than a vegetative clump of individuals. We found three large clumps in 16 hectares (Fig. 3). Two of the three large clumps are associated with low, wet areas. However, the other large clump was found on the side of an esker, which is rather dry. The size and distribution of these clumps may infer the history or invasion of the big woods plot, and also the future of *B. thunbergii*’s invasion.

It is not surprising that *B. thunbergii* has colonized the Big Woods plot at the George Reserve as it is in an area near former agricultural sites. *B. thunbergii* is more successful at invading abandoned agricultural sites, especially before a canopy has been established (DeGasperis and Motzkin, 2007). It could be that *B. thunbergii* initially colonized less-shaded areas of the woods, such as the esker, which is where the main road is located. Once established, it could invade the rest of the forest, and flourish in low, wet areas, where we see it thriving today.

These large, dense clumps of *B. thunbergii* are probably growing in size, but have not yet taken over the forest floor. Only 3.1% of the forest floor is covered by more than 80% *B. thunbergii*, and 15.6% is covered by 60-79%. It will be interesting to see if the *B. thunbergii* population will grow over time, or if it will be limited by water availability. It is likely to succeed in invading, because *B. thunbergii* can increase the level of nitrogen in the soil (Ehrenfeld et al, 2001) its most limiting nutrient for growth (Cassidy et al, 2004).

If the *B. thunbergii* in the George Reserve is not limited by water, and can increase its invasion by altering soil-based ecosystem processes to dominate the understory, it may sow its own seeds of destruction. Such a high density of *B. thunbergii* may institute a classic Janzen-Connell effect. These large, dense clumps may attract herbivores that completely defoliate *B. thunbergii*. Such defoliation was first documented by Neiswander in 1941, and has been recently documented in New Jersey, by the native moth *Coryphista meadii* (Ehrenfeld, 2009). While reports of *C. meadii* in the region are few and far between, populations have been found in Ohio (Williams et al. 1997). It is possible that *C. meadii* could extend its range to Michigan and the George Reserve, potentially impacting the *B. thunbergii* population in the Big Woods.

It is important to note that *B. thunbergii* is not the only invasive shrub in the George Reserve. *Elaeagnus umbellata* is also edging its way into the forest, after its primary invasion of fields and roads surrounding the Big Woods plot. *E. umbellata* has created low, dense canopies with the help of other sub-canopy species such as *Hamamelis virginiana*. While we did not find a negative correlation between the presence of *H. virginiana* and *B. thunbergii*, we did observe in
the field that under low, dense canopies there was less *B. thunbergii*. This may already be limiting the invasion of *B. thunbergii*, and would be an interesting follow-up study to conduct. Future studies and surveys are needed to see if *B. thunbergii* will be limited by low water availability throughout the Big Woods. While the Big Woods is located within a reserve, in other natural areas control of *B. thunbergii* could potentially be achieved by the introduction of *C. meadii* or the maintenance of a thick understory. Only future surveys and studies will answer the question of how and why *B. thunbergii* is limited in distribution, or if it is at all.

ACKNOWLEDGEMENTS

We would like to thank the E.S. George Reserve and David Allen.

LITERATURE CITED


THE IMPACT OF HABITAT TRANSITIONS AND EDGE EFFECTS ON HICKORY-FEEDING LEAF MINERS

SUSAN CHENG, CINDY BICK, ANA CHARA, DAVID GONTHIER, MINGQI LIU, WILLIAM WEBB, IVETTE PERFECTO

Abstract. Environmental conditions, such as habitat openness and light availability, can affect leaf miner populations. To determine the impact of a habitat transitioning from an open field to a forest, the population characteristics of two leaf miners, *Liriomyza brassicae* and *Phyllonorycter elmaella*, found on *Carya* spp. seedlings were examined. The abundance of leaf miners per plant was seen to have a positive relationship with plant height (*P. elmaella*: $p < 0.001$; *L. brassicae*: $p < 0.001$). Results also showed that the number of *P. elmaella* per plant decreased with distance from the edge of the open field ($p < 0.05$). Mortality for *P. elmaella* also decreased moving farther from the open field and closer to the forest, although this trend was not statistically significant. No trend was observed between distance and *P. elmaella* mine lengths. *L. brassicae* mine lengths increased when moving away from the edge of the open field and towards the forest. No trend between abundance of *L. brassicae* and distance or mortality along the habitat transition was observed, suggesting this species of leaf miner does not display a habitat preference.

INTRODUCTION

Many abiotic and biotic factors affect the distribution and abundance of species (Hairston *et al*. 1960). Specifically, insect herbivores may be influenced by the presence of predators and parasitoids (Hairston *et al*. 1960) or plant traits, such as plant defenses and nutritional status (Awmack & Leather 2002). Leaf miners are particularly sensitive to environmental and habitat characteristics, including light availability and habitat openness (Marino *et al.* 1993, Potter 1992). As such, habitat edges can strongly influence leaf miner abundance and mortality. A previous study has shown that an Agromyzidae leaf mining fly was more abundant near edges of forest habitats while its overall fitness and survivorship were lower at the edge as compared to within forests (McGeoch & Gaston 2000). Since leaf mining herbivores are important pests of many crops, such as coffee (De la Mora *et al*. 2008) and tomato (Wolfenbarger & Wolfenbarger 1966), and can reduce the growth or fitness of their host plants (Norris 1997; Thalmann *et al*. 2003), understanding how habitat transitions and edges influence leaf miners can shed insight on how habitat conditions can control insect populations.

In eastern Michigan, two leaf miners can be commonly observed on hickory seedlings (*Carya* spp; *C. cordiformis, C. Glabra, C. Ovata, C. laciniosa*). The first is *Liriomyza brassicae*, a dipteran that forms long narrow mines. *L. brassicae* adults oviposit eggs on host leaf surfaces, and when the larvae hatch after three days, they enter the mesophyll and consume leaf tissues in a long and narrow tunnel. Approximately a week later, the larvae will emerge from the leaf and undergo pupation on the ground for two weeks before the adults emerge (Beri 1974). The second leaf miner, *Phyllonorycter elmaella*, is a Lepidopteran miner that has a similar life history of adult oviposition and larvae consumption (Bernardo *et al*. 2006). In comparison to *L. brassicae, P. elmaella* mines are blotchy and wide in appearance.
The purpose of this study was to determine whether the transition between an oak-hickory forest and an open field would impact the population characteristics of *L. brassicae* and *P. elmaella*. For both leaf miners, it was predicted that 1) abundance of leaf miners would have a negative relationship with distance from the edge of the open field, 2) mine length would be positively correlated with distance and canopy cover, and 3) mortality would be greater towards the edge of the open field and smaller within the forest.

**METHODS**

This study was conducted at the E. S. George Reserve in Livingston County, MI at the transition between an open field and a forest. Four 50 meter (m) transects were laid northwest starting from the edge of the open field and ending in a forest. At each transect, *Carya* spp. seedlings less than 1 m in height and within 0.5 m of either side of the transect line were counted. For each *Carya* spp. seedling, the distance of the plant from the edge of the open field, the height to the apical meristem, and the canopy cover were measured. Canopy cover was estimated by taking five densitometer readings at each cardinal direction and above the seedling, resulting in a canopy cover range of 0, 20, 40, 60, 80, and 100 percent.

For each *Carya* spp. seedling, the number of *L. brassicae* mines and *P. elmaella* mines were counted. Information on whether the leaf miners were alive (moving within the mine), emerged (exit hole of mine was present and no dead larvae were found) or dead (remains of larvae observed) as well as the length of each mine was collected. The length of each mine was measured by tracing the mine with a string and then measuring the length of the string.

R, SPSS, PAST, and Excel were used to analyze the data using linear regressions and to calculate p values. Measurements for each distance point along the four transects were used to determine the relationship between distance and leaf miner population characteristics. Data from the four transects were also converted into plots (1 m by 10 m) and used to compare leaf miner population characteristics with one-way ANOVA. In cases of non-normal distributions, data were analyzed with Kruskal-Wallis non-parametric tests (SPSS).

**RESULTS**

*Canopy Cover and Plant Height*

As distance from the edge of the open field increased, % canopy cover increased (p < 0.001) (Figure 1). However, there was a negative relationship between *Carya* spp. seedling height and distance from the edge of the open field (p < 0.001) (Figure 2).
Abundance

There was no trend between the abundance of *L. brassicae* per plant and distance from the open field’s edge (Figure 3). On the other hand, the abundance of *P. elmaella* per plant did decrease with distance from the open field’s edge (p<0.05) (Figure 4).

In addition, the abundance of the two leaf miner species had a statistically significant positive relationship with plant height (*P. elmaella*: p < 0.001; *L. brassicae*: p < 0.001) (Figure 5 and 6).
Mine Length

As distance from the open field’s edge increased, the average length of the *L. brassicaceae* mines increased (p < 0.001) (Figure 7). No trend was observed between distance and the *P. elmaella* mine lengths (Figure 8).

Figure 7: Average length of *L. brassicaceae* mines.
Figure 8: Average length of *P. elmaella* mines.

\[ y = -0.0079x + 3.5888 \]

\[ R^2 = 0.0016 \]

**Mortality**

No *L. brassicae* were found alive and there was no trend between dead *L. brassicae* and distance. The number of alive *P. elmaella* per plant tended to increase when moving farther from the open field’s edge and moving closer to the forest. However, this relationship was not statistically significant (p>0.05) and was not observed when analyzed by distance intervals. The proportion of dead *P. elmaella* to total leaf miners seemed to decrease with distance from the edge, but was not statistically significant (p>0.05) (Figure 9).

Figure 9: Mortality levels of *P. elmaella* along distance intervals from the open field’s edge.
DISCUSSION

*L. brassicae*

Although *L. brassicae* and *P. elmaella* exhibit similar life history, these herbivores display different foraging strategies (Beri 1974 and Barrett & Brunner 1990). Unlike previous studies by Ong et al. (2009), *L. brassicae* was abundant at forest and open field edges as well as in areas near and within the forest. This suggests that *L. brassicae* does not exhibit any habitat preferences for forest edges or the deep forest. However, our study site had an even distribution of hickory seedlings at the edges and towards the deep forest. Rickleffs and Miller (1999) stated that plant hosts in homogenous communities are easier for herbivores to find compared to plant hosts in heterogeneous communities. Therefore, it is possible that *L. brassicae* was abundant along the forest gradient because hickory seedlings were easier to locate.

The length of *L. brassicae* mines increased as the distance along the forest gradient increased, which was consistent with previous studies by Ong et al. (2009). Rickleffs and Miller (1999) concluded that not only do herbivores concentrate in certain habitats, but their enemies will also. On average, *L. brassicae* mines were shorter in open areas compared to areas closer and within the forest. Predation by parasitic wasps may account for the decreased mine lengths (Barrett & Brunner 1990). Additionally, predators may concentrate on forest edges due to high light levels; thus making the leaf miners easier to find.

However, distance from the open field’s edge was not related to mortality. Despite the uniform distribution of hickory seedlings throughout our study site, seedling density seemed to decrease as we moved deeper into the forest. Perhaps adult *L. brassicae* are not concentrated within the forest areas because plant hosts are limited. Since *L. brassicae* show no habitat preference, it is also possible that it expends a lot of energy through high reproductive effort to ensure some offspring success in areas with high predation risk.

*P. elmaella*

A previous study (Ong et al. 2009) suggested that *P. elmaella* did not have any habitat preferences. On the contrary, our study suggests that the abundance of *P. elmaella* was inversely proportional to the distance from the edge. The site of oviposition is a function of parental choice and various organisms discriminate between oviposition sites based on the availability of resources and the optimization of offspring success (Resatarits 1996, Kiesecker and Skelly 2000). Mines found at higher densities at the edges suggest some inherent value in ovipositing in these areas compared to ovipositing deeper in the forest. Hickory seedling density seemed to decrease as the distance increased towards the forest, and therefore adult *P. elmaella* may choose to oviposit their eggs in areas with high density of hickory seedlings.

The relationship between the length of *P. elmaella* mines and the distance from the forest edge could not be determined. Perhaps there are other generalist parasitoid wasps attacking *P. elmaella*, which could account for the differences in mine length throughout the forest. However, *P. elmaella* mortality decreased with distance from the forest edge. Since canopy cover was highest in the forest compared to its edges, *P. elmaella* larvae may be protected from parasitoid wasps when in the forest.
AKNOWLEDGEMENTS

We would like to thank Ivette Perfecto, John Vandermeer, and Dave Allen for scientific inspiration and coordination of Field Ecology.

LITERATURE CITED


ABSTRACT

During this study we observed seedling distribution and abundance in order to gain insight into whether it could provide clues into possible successional shifts in vegetation occurring in a Michigan forest. With previous data on the mature trees in the Big Woods of the E.S. George reserve, we investigated seedling distribution within inter-facing clumps of black cherry (*Prunus serotina*), red-maple (*Acer rubrum*) and (*Hamamelis virginiana*). We predicted that the seedling composition would mirror that of its respective overstory, with diminishing colonization by multiple species as the canopy became more homogenous along the transects. Statistical analyses show that three out of sixteen transects showed differences in successional dynamics when comparing the seedling and adult communities. In 13 out of 16 transects sampled, we found a significant difference between the seedling and overstory species compositions.

INTRODUCTION

Many North American temperate forests are undergoing a successional shift in species composition. Some forests are recovering from logging activities, while others are changing due to climate change, disease, or invasions from exotics. Understanding and attempting to predict these shifts in composition is important for productive forest management.

Anthropogenic disturbances, specifically controlled burning, helped maintain oak forests from succession by other tree species such as Red maple. Red maple has increased after a wide range of disturbances and as a mid-successional species in many forest types (Abrams 1998). The shift from primary (oak) to secondary successional species (red maple) is seen in shade forest communities. Maple can regenerate in the shade of the canopy trees whereas oaks cannot compete in the their own shade. The unique ability of red maple to behave as both an early and a late successional species and to thrive in widely varying edaphic conditions has therefore promoted its increase in both disturbed and fire-suppressed landscapes (Abrams 1998). The increase in red maple and decrease in oak in eastern forests during the twentieth century may be explained, in part, not only by forest fire suppression and landscape disturbance, but also by the differential impacts of deer.

The Big Woods is located in the Edwin S George Reserve and is approximately 18 hectares (400x500). Black Cherry *Prunus serotina*, Red Maple *Acer rubrum*, and Witch Hazel
Hamamelis virginiana are three tree species located in the mid and sub-canopy. We examined these species spatial patterns in eight locations within the Big Woods. These species are predominately found in clumps. The overstory of the Big Woods is mostly dominated by oak trees, yet the understory shows a rising dominance of Red maple, Black cherry, and Witch hazel seedlings. Such a trend seems to follow congruently with the majority of North American forests, which were once managed by fire, yet now are experiencing species shifts given that lack of these suppression management practices.

The purpose of this study is to determine seedling distribution within the sub canopy among the Black Cherry Prunus serotina, Red Maple Acer rubrum, and Witch Hazel Hamamelis virginiana clumps in the Big Woods. By comparing the relative species abundance of the seedlings on the forest floor and whether they are proportional to the overstory canopy distribution we might be able to see evidence of an ongoing species shift. Measuring relative abundance of seedling in the understory this will assist in the prediction of the dominant succession species. We believe that the seedling distribution mimics the clump species indicating no change in the clump species.

METHODS

We conducted our study in Big Woods Forest, found in the NE corner of the E.S George Reserve. In 2007/2008, Allen et. al performed a census and found Acer rubrum (Red maple), Prunus serotina (Black cherry), and Hamamelis virginiana (Witch hazel) to comprise majority of the mid-canopy in a clumped distribution. To test if these species clumps were potentially increasing or decreasing in size or concentration or mixing randomly, we analyzed seedling distribution and compared it to the canopy above.

We ran eight non-random 2 x 50 m transects. Six transects (Figure 1) passed from one species clump to another, two of each of the following: Red maple to Witch hazel, Witch hazel to Black cherry, Black cherry to Red maple, and two transects through forest sections without clear species clumping. Using the Allen et. al distribution map as reference, we walked to the approximate boundary between species clumps and ran the transect 25m perpendicular in each direction. Untagged individuals of our target species were counted for each meter increment and categorized by height (0-25cm, 25-50cm, 50-75cm, 75-100cm, >1m). We recorded only living individuals (i.e. with leaves) with majority of their stem(s) inside the transect boundary. We regarded clonal stems as one individual, and used the highest stem as the sole height measurement.
To map the transition between species clumps in the canopy, a 10x50 m transect of adult trees was overlain above the understory transects. Data from both canopy and ground transects were divided evenly, and seedling/seedling and seedling/canopy percentages were analyzed using a chi-squared test.

Figure 1: Map of George reserve’s Big Woods plot with the transects sampled for this study

RESULTS

Chi-square analyses were conducted on the seedling community to compare the proportion of individuals of the three main species investigated on either side of each transect surveyed. P-values indicate that seven out of eight transects were not similar in species composition as the species found on the second half of the transects compared (Table 1). Integration of each species into patches of the remaining two was compared. Only one of the two transects surveyed for red maple/witch hazel was shown to not be significant at 0.946 indicating that the transect did not sample in a region in which a species transition, or patch, was occurring. This was attributed to sampling error, which was confirmed when the transects were mapped according to GPS data of the adults from previous samplings conducted at the plots (Allen 2010, personal communication).
Table 1: P-values obtained of the chi-square analysis comparing seedling composition on either side of the transects.

<table>
<thead>
<tr>
<th>Transect</th>
<th>p-values</th>
<th>significant?</th>
<th>difference</th>
</tr>
</thead>
<tbody>
<tr>
<td>BC/RM #1</td>
<td>7.48E-12</td>
<td>Y</td>
<td>more bc less rm</td>
</tr>
<tr>
<td>BC/RM #2</td>
<td>1.37E-07</td>
<td>Y</td>
<td>more rm</td>
</tr>
<tr>
<td>RM/W #1</td>
<td>1.27E-03</td>
<td>Y</td>
<td>less bc more wh</td>
</tr>
<tr>
<td>RM/W #2</td>
<td>9.46E-01</td>
<td>NS</td>
<td>expected</td>
</tr>
<tr>
<td>W/BC #1</td>
<td>1.83E-13</td>
<td>Y</td>
<td>less bc, wh more rm</td>
</tr>
<tr>
<td>W/BC #2</td>
<td>1.32E-03</td>
<td>Y</td>
<td>less bc more wh</td>
</tr>
<tr>
<td>Mix #1</td>
<td>1.98E-02</td>
<td>Y</td>
<td>more bc less rm</td>
</tr>
<tr>
<td>Mix #2</td>
<td>2.37E-03</td>
<td>Y</td>
<td>more bc less rm</td>
</tr>
</tbody>
</table>

Another set of Chi-square analysis were conducted to compare the seeding community composition to the upperstory adult community of the three main species investigated for each transect surveyed. P-values indicate that three out of sixteen transects were not similar in the seedling species community in comparison to the upperstory community (Table 2). There was no relationship between the transects that did mirror the upperstory community composition to those that did not.

Table 2: P-values obtained of the chi-square analysis comparing seedling and adult community composition.

<table>
<thead>
<tr>
<th>Transect</th>
<th>p-value</th>
<th>Significant?</th>
<th>difference</th>
</tr>
</thead>
<tbody>
<tr>
<td>BC/RM #1</td>
<td>0.086</td>
<td>NS</td>
<td>expected</td>
</tr>
<tr>
<td>BC/RM #1</td>
<td>0.016</td>
<td>SIG</td>
<td>more rm less bc</td>
</tr>
<tr>
<td>BC/RM #2</td>
<td>2.96E-28</td>
<td>SIG</td>
<td>more wh less rm</td>
</tr>
<tr>
<td>BC/RM #2</td>
<td>2.11E-10</td>
<td>SIG</td>
<td>less bc more wh</td>
</tr>
<tr>
<td>RM/W #1</td>
<td>2.32E-11</td>
<td>SIG</td>
<td>less bc more rm</td>
</tr>
<tr>
<td>RM/W #1</td>
<td>7.75E-14</td>
<td>SIG</td>
<td>less bc more rm</td>
</tr>
<tr>
<td>RM/W #2</td>
<td>1.48E-03</td>
<td>SIG</td>
<td>more rm</td>
</tr>
<tr>
<td>RM/W #2</td>
<td>0.681</td>
<td>NS</td>
<td>expected</td>
</tr>
<tr>
<td>W/BC #1</td>
<td>4.87E-07</td>
<td>SIG</td>
<td>more rm</td>
</tr>
<tr>
<td>W/BC #1</td>
<td>4.49E-16</td>
<td>SIG</td>
<td>less bc more rm</td>
</tr>
<tr>
<td>W/BC #2</td>
<td>0.016</td>
<td>SIG</td>
<td>less bc more rm and wh</td>
</tr>
<tr>
<td>W/BC #2</td>
<td>0.005</td>
<td>SIG</td>
<td>less bc more rm and wh</td>
</tr>
<tr>
<td>Mix #1</td>
<td>7.55E-05</td>
<td>SIG</td>
<td>less bc more wh</td>
</tr>
<tr>
<td>Mix #1</td>
<td>1.22E-04</td>
<td>SIG</td>
<td>less bc more wh</td>
</tr>
<tr>
<td>Mix #2</td>
<td>0.011</td>
<td>SIG</td>
<td>less bc more rm</td>
</tr>
<tr>
<td>Mix #2</td>
<td>0.164</td>
<td>NS</td>
<td>expected</td>
</tr>
</tbody>
</table>
DISCUSSION

Statistical analyses confirm that there is a change currently occurring in the species community. This has been apparent since the last sampling conducted by Allen of the adult tree community in the plots. This investigation showed that the succession of the three main species occurring are still undergoing with red maple as the forerunner of the three species. Investigating the seedling community allowed us to associate the distribution and abundance of seedlings as a characteristic of the relative fecundity of each species when compared to one another. Literature confirms that red maple has high success rates of seed dispersal along with early reproductive maturity help it out compete non-opportunistic neighbors (Burns and Honakala 1990).

Red maple, considered a “super-generalist,” exhibits traits that allow it to succeed in a wide range of successional stages (Abrams 1998). These traits allow this species to succeed in low quality environments characterized by low soil nutrient and moisture levels as well as highly disturbed regions give this species an overall advantage over native species (Kloeppe1 et al. 1993 and Abrams 1998). As more investigations have been done regarding this trend of succession across many forests throughout the nation genetic analysis has been used as a resource to understand other intrinsic factors that might be helping drive these successions at the molecular level. Researchers have realized that red maple, on top of having physiological and reproductive advantage over other natives also possess an affinity for hybridization with other related species (Abrams 1998).

Practices conducted by the previous managers of these lands, scheduled burns, were enough to suppress species like Red maple from dominating oak forests. With changed in human demographics and shifts in anthropogenic methodologies the end result seems to be leading in the direction of many forests becoming monocultures of highly opportunistic ruderal species that are able to outcompete and sustain their population numbers even in the face of rapid abiotic changes and stress. The challenge now becomes finding a way to predict the patterns of these ongoing changes and how can we use such information to help maintain some of the species that fall vulnerable to opportunistic species.

For future research it is suggested that continuous sampling be done of both the understory and the seedling communities in order to connect any trends that might not be apparent from simply investigating the community dynamics of the upper canopy. With these empirical data, powerful models can be formulated to help predict approximate patterns and reveal areas of vulnerability that these dominating species might possess, yet might not be apparent when observed in the timescale of an individual lifetime. With these predictions, land managers might formulate management techniques that can once again manage these overbearing successional trends, without much disturbance to the surrounding fauna communities.

ACKNOWLEDGEMENTS

We would like to thank John Vandermeer and Ivette Perfecto for their advice.
LITERATURE CITED


Allen, D. Personal communication. 2010


NESTING SITE LIMITATIONS AND NESTING RESOURCES OF CAVITY NESTING ANTS (HYMENOPTERA: FORMICIDAE) IN THE E. S. GEORGE RESERVE, MI, USA.

CINDY BICK, BINBIN LI, MINGQI LIU, RAFAEL D’ANDREA, KATHERINE ZEMENICK, STACY PHILPOTT

ABSTRACT

The stability and function of ecosystems are maintained by species composition and diversity. However, very little studies have examined the availability and limitation of potential cavity nesting sites in ant communities. Nesting cavities were collected from 30 plots in a predominately oak-hickory forest at the E. S. George Reserve on September 25, 2010. A total of 330 hickory seeds and acorn nuts were collected and identified according to species. The nests were assessed for fungi, frass and the presence of ants. Nests were opened to calculate the proportion occupied by cavity-nesting ants. Across all plots, a total of 3 ant species were found. The most common ant species found was Leptothorax curvispinosus (n = 11). Myrmica punctiventris (n = 4) and Tapinoma sessile (n = 1) were not as common. Additionally, all samples of M. punctiventris was found in one plot. Nest cavity species were homogenous across all plots, with Quercus spp. and Carya spp. as the most common. Ant preference for a specific nesting cavity species or size could not be determined. There did not seem to be a relationship between the availability of nests and the diversity of ant species found, as no plots had more than one ant species present. Additionally, ant occupation of nesting cavities with frass and fungal growth was inconclusive. Expansion on the size of the plots as well as an equal number of available nesting species may provide more information on ant community assemblages and ant species nesting preferences.

INTRODUCTION

Previous studies have shown that the stability and function of ecosystems are maintained by species composition and diversity (Schulze & Mooney 1993, Tilman & Downing 1994, Naeem et al. 1994, Tilman et al. 1996, Hooper & Vitousek 1997, van der Heijden 1998). However, the ecological mechanisms that maintain and regulate biodiversity are not well understood. Several mechanisms such as diet specializations, habitat selection, spatial or temporal resource partitioning, and competitive interactions have been proposed to explain coexistence in animal communities which gives rise to species richness and abundance. Numerous studies have examined these dynamics in insect communities (Heinrich 1976, Davidson 1978, Whitman 1980, Thomson 1981, Dyer & Floyd 1993, Denno et al. 1995, Barone 1998, Chikatunov et al. 1998). However, relatively few studies (Longino 1989, Friedrich & Philpott 2009) have examined the ecological mechanisms that maintain and regulate biodiversity in ant communities; particularly on specific resource availability and limitations.

Ants interact among different functional groups of organism that constitutes ecosystems. Some species of tropical ants are considered keystone species that act as agricultural pest control agents and influence the surrounding biotic and abiotic communities (Perfecto & Vandermeer 2006, Liere & Perfecto 2008). Philpott and Foster (2005) reported that ants are the most
numerous macro-invertebrates and account for at least half of the animal biomass present in most ecosystems. They occur in a wide array of habitats and play various ecological functions as seed dispersers, soil aerators, and predators of other insects (Hollodobler & Wilson 1990). These ecological functions are critical to forest dynamics. They also nest in a variety of resources such as fallen logs, leaf litter, bark, sticks and twigs as well as the cavities of acorns and nuts (Hollodobler & Wilson 1990).

Their range of nesting preferences and behavior is a further reflection of ant species diversity. Some ants do not have a specific preference for a particular nesting resource while others nest solely in cavities and are highly dependent on the availability of those cavities for the survival of their colonies (Herbers 1986). Cavity dwelling ants are nest site limited and interspecific and intraspecific competition for nesting resources is intense (Ambrecht et al. 2004). Therefore, cavity nests are extremely important to cavity nesting ants and the limitation of nesting resources may have a profound effect on ant diversity. The reduction in numbers of nest sites has been reported as a mechanism causing ant diversity loss in disturbed ecosystems (Philpott & Foster 2005, Friedrich & Philpott 2009).

Our study focused solely on hickory nuts and oak acorns as potential nesting sites for ants because they were the most common tree species present at our sites. Seven cavity nesting species of ants have been found at the E. S. George Reserve (Larsen et al. 2008). These include: 3 different species of Leptothorax, Myrmica punctiventris, Ponera pennsylvanica, Lasius alenius, and Tapinoma sessile. The mechanisms that maintain and regulate biodiversity in cavity nesting ants was assessed by examining: (1) the availability and limitation of potential cavity nesting sites, (2) the preferences of different species of ants, and (3) how ant communities may be assembled. The increasing numbers of available nests provides additional and more diverse nesting resources that may lead to an increased diversity of ants. Ants may also differ in niches, and thus diverse nesting resources might also produce diverse ant assemblages.

METHODS

The study was conducted on September 25th at the ES George Reserve, in the Big Woods Plot. The Big Woods is a oak-hickory forest, with the understory dominated by Acer rubrum, Prunus serotina, and Hammamelis virginiana. We placed three 100m transects along three different roads within the Big Woods Plot. At 10m intervals, we placed a 0.5 m² quadrat and collected all nuts and acorns within it. A stopwatch was used to randomly select the distance of the quadrat from the road; the hundredth of a second designated the number of meters we traveled into the forest from the edge of the road.

All of the hickory nuts and oak acorns from each plot were measured, identified, and opened to determine the presence of an ant colony, fungal growth, beetle larvae and adult, and beetle frass. Hollow nests were considered available nests. When ants were present, they were collected and preserved in alcohol for later identification under a microscope. The Ants of Ohio dichotomous key was used to identify the different ant species. Regression analyses were used to determine the relationship between available nesting sites, frass, fungal growth on nests, and the presence of ants in different nuts. The Spearman Rank correlation analysis was used to determine ant preferences with respect to different acorn and nut sizes.

RESULTS

A total of 330 hickory nuts and oak acorns were collected from 30 plots on the E. S. George Reserve. Only three species of ants were found: Leptothorax curvispinosus, Myrmica
*punctiventris* and *Tapinoma sessile*. *L. curvispinosus* occupied all three common nesting species with the most ants found in *Carya glabra* (Figure 1). All colonies of *M. punctiventris* were found in the same plot, which consisted only of *Quercus spp.* of nests. Only one individual of *T. sessile* was found.

The hickory nuts and oak acorns were divided into 15 different size categories of 1000 mm\(^3\) increments. Individuals within one colony of *M. punctiventris* were found in different nut sizes. However, *Leptothorax curvispinosus* did not demonstrate any preference for a particular nest size as it was found throughout different size gradients (Figure 2). The correlation between nest size and the presence of ants was not significant (p = 0.897). The data for *M. punctiventris* and *T. sessile* was excluded to test the correlation between the availability of hickory nests and the presence of *L. curvispinosus* in these nests. However, the Spearman Rank correlation analyses showed no correlation between the availability of hickory nests and the presence of ants (Table 1).

![Fig. 1. Proportion of acorns/nuts and ant species found.](image)

Due to the limited scale of the experiment and the paucity of ant colony samples, the only species for which an attempt at estimating preference could be made was *L. curvispinosus* (11 colonies were sampled). Of the three categories of nest species available, the most common one found in the plots were the *Quercus spp.*. The sample sizes for the other two species were small and, therefore ant preferences for certain species of nesting cavities could not be determined (Table 2).

![Fig. 2. Number or ant species found within different sizes of nuts.](image)
Table 1. Results of Spearman Rank correlation analysis

<table>
<thead>
<tr>
<th>Factor</th>
<th>Total nuts</th>
<th>Hollow nuts</th>
<th>Hollow Nuts fraction</th>
</tr>
</thead>
<tbody>
<tr>
<td>All Species</td>
<td>P value</td>
<td>.260</td>
<td>.101</td>
</tr>
<tr>
<td></td>
<td>R^2</td>
<td>.280</td>
<td>.399</td>
</tr>
<tr>
<td>Factor</td>
<td>Oak nuts</td>
<td>Oak Nuts fraction</td>
<td>Hollow Oak nuts</td>
</tr>
<tr>
<td>All Species</td>
<td>P value</td>
<td>.480</td>
<td>.582</td>
</tr>
<tr>
<td></td>
<td>R^2</td>
<td>.198</td>
<td>-.139</td>
</tr>
<tr>
<td>Factor</td>
<td>Hickory nuts</td>
<td>Hickory nuts fraction</td>
<td>Hollow Hickory nuts</td>
</tr>
<tr>
<td>Leptothorax curvispinosus</td>
<td>P value</td>
<td>.384</td>
<td>.356</td>
</tr>
<tr>
<td></td>
<td>R^2</td>
<td>.292</td>
<td>.247</td>
</tr>
</tbody>
</table>

All hollow nests were considered available nesting sites for ants. The high correlation between frass and hollow nests was a consequence of regarding every nest with frass as hollow (Figure 3). Additionally, ants did not seem to choose nesting sites based on the size of the nest. Their distribution among the different size categories was relatively the same (Figure 3). Furthermore, the presence of fungi in a nesting cavity did not correlate with the size of the hollow nests. Nests with fungal growth were not correlated to the presence of ants in a nest (Figure 4).

Fig. 3. The distribution of hollow nests, nests with frass, nests colonized by fungi, nests occupied with ants, sorted by size.
Ants also seem to show no preference for a specific species of nesting cavity (Table 2). No evidences of nest limitation and interspecific competition in ants were found. The diversity of available nesting sites was mostly homogenous with one species found in most plots. None of the plots had more than one ant species colony.

Table 2. Data on nut occupancy sorted by genus. *Note: all colonies of M. punctiventris were found in the same 0.5m X 0.5m quadrat. This might indicate that this is actually one data point and not 4. All hollow nuts found in that plot were oak acorns.

<table>
<thead>
<tr>
<th>Nest found</th>
<th>%</th>
<th>frass</th>
<th>%</th>
<th>fungi</th>
<th>%</th>
<th>ants</th>
<th>%</th>
<th>L. curvispinosus</th>
<th>M. punctiventris</th>
<th>T. sessile</th>
</tr>
</thead>
<tbody>
<tr>
<td>Oak</td>
<td>109</td>
<td>0.52</td>
<td>66</td>
<td>0.61</td>
<td>14</td>
<td>0.13</td>
<td>8</td>
<td>0.07</td>
<td>3</td>
<td>4*</td>
</tr>
<tr>
<td>Hickory</td>
<td>89</td>
<td>0.43</td>
<td>41</td>
<td>0.46</td>
<td>23</td>
<td>0.26</td>
<td>8</td>
<td>0.09</td>
<td>8</td>
<td>0</td>
</tr>
<tr>
<td>Other</td>
<td>10</td>
<td>0.05</td>
<td>2</td>
<td>0.2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Total</td>
<td>208</td>
<td>109</td>
<td>37</td>
<td>16</td>
<td>11</td>
<td>4</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**DISCUSSION**

In contrast to previous studies (Byrne 1994, Philpott & Foster 2005, Friedrich & Philpott 2009), the sites at E. S. George Reserve do not seem to be nest site limited. Additionally, the presence of ants may not vary directly with available nest abundance. Evidence of competition for nesting space between ant species could not be determined due to the limited amount of ant colonies sampled. Overall, the relationship between the number of available nests (hollow nests)
and the number of ant species found in a plot was inconclusive. Although four colonies of *M. punctiventris* were found, all samples were from the same plot. It is likely that all occupied nests in the same plot were buds from the same colony. Only one colony of *T. sessile* was found and therefore, its preference for nest size and species of nest could not be determined. Ants did not seem to differentiate between nests with frass and fungal growth, even at different nest sizes.

The study by Larsen et al. (2008) was conducted during a masting year. Their assumption was that ant occupancy was low due to the superabundance of available hickory nuts and oak acorns. However, our study was conducted during a non-masting year. The percentage of ant occupancy was low and the superabundance of available nesting sites was also observed. It is possible that during a masting year, more hickory nuts and oak acorns are added to a system that is already abundant in hickory nuts and oak acorns. It is also possible that ant diversity has also decreased because *L. curvispinosus* and *M. punctiventris*, both generalist species; were commonly found. Another possibility is that these two species are still active at cooler temperatures and therefore, more common during the fall when sampling for both our study and the Larsen et al. (2008) study was conducted.

The species of nests found in each plot had very little variability and none of the plots had more than two ant species present. Additionally, there did not seem to be intense competition for nesting resources because less than 5% of the total number of hickory nuts and oak acorns were occupied by ants. A percent occupancy of less than 30% was also observed in previous studies at the E. S. George Reserve (Larsen et al. 2008). In contrast to our study, Larsen et al. (2008) reported a higher percentage of occupation by a single species in the total available nests. Furthermore, Larsen et al. (2008) collected a hundred more hickory seeds and acorn nuts and used a 1 m² plot. In future research, the size of the plots should be expanded to further elucidate ant community assemblages. An equal number of hickory nuts and oak acorns should also be collected in order to assess ant nesting preferences.

**REFERENCES**


SOURCE-SINK DYNAMICS OF *PEROMYSCUS LEUCOPUS* POPULATION IN THE GEORGE RESERVE

ANA CHARÁ, SAHAR HAGHIGHAT, JOHN GUITTAR, KASSANDRA C. SEMRAU, LAUREN MALONEY, RACHEL CABLE, PHIL MYERS

ABSTRACT

The white-footed mouse, *Peromyscus leucopus*, has been known as an abundant species on the E.S. George Reserve for many years, despite fluctuations in population abundance from year to year. In addition to surveying the population abundance through live trapping techniques as in years past, we also compared the populations between two habitats, field and forest, to determine whether the source-sink model applies to the current population of *P. leucopus* in the George Reserve. After comparing the size and reproductive stages of both males and females between the two habitats, we found that females did tend to follow the source-sink model by residing in the resource-rich forest when fit and reproductive and residing in the field when younger and not yet reproductive, while males did not follow this model.

INTRODUCTION

*Peromyscus leucopus*, the white-footed mouse, has been the most abundant rodent in the E. S. George Reserve over the past 10 years (Journal of Field Ecology 1998-2000, 2006-2009). *P. leucopus* serves an array of ecological functions in temperate forests, including primary seed dispersal, seed predation (Wolff 1984), and as a winter food source for predatory mammals and birds (Meyer, pers. comm. 2010). In addition, *P. leucopus* is a vector for the botfly parasite as well as a reservoir host for Lyme disease (Ostfield et al. 1996).

As global temperatures increase in response to global climate change, many temperature-dependent mammals are driven to higher latitudes. Habitat fragmentation and habitat alteration further affect mammal distributions by depriving or providing artificial habitat. It is important to study mammal ranges and habitat preference over time, as shifts in interactions and/or competition can have strong direct and indirect effects with environmental processes (Ostfield et al. 1996).

For small mammals the availability of suitable habitat is an important determinant of their distribution in the forest. Several authors have found that the number and diversity of small mammal captures depend strongly on ecological factors such as food availability, temperature, predation risk, interspecific competition, nesting and roosting sites (Sato 2007).

Territoriality is an important factor in *P. leucopus* spatial distribution; they have a well developed homing behavior with home ranges average of 0.1 ha. Females are territorial during the breeding season and their home ranges overlap with those of the males (Linzey and Brecht, 2005).
Here, we report on small mammal capture rates at the E. S. George Reserve. We also investigate whether the distribution of large populations of *P. leucopus* follows a source-sink model that has been proposed for other wood mouse species (Tattersall et al. 2004). That is, fit adult *P. leucopus* competitively dominate the ideal habitat - in our case the forests with greater food and shelter - and juvenile or un-fit individuals are relegated to less-ideal habitats - in our case the field where mice are more susceptible to predation and food scarcity. In this study, we test the hypothesis that larger reproductively active adult *P. leucopus* are found in the superior forest habitat.

**MATERIALS AND METHODS**

On September 25th from 15:00-17:00, a total of 200 traps were set out along six transects (two in the field and four in the forest) throughout the E.S. George Reserve. Sixty-five small (5.4 x 6.3 cm x 16.5 cm) Sherman traps were set along field transects and 135 large (8 x 9 x 23.5 cm) Sherman traps were set along forest transects. Whole rolled oats were used as bait and traps were placed approximately 10-15 meters apart in the field and 15-20 meters apart in the forest. Locations were chosen based on where small mammals were likely to be found or forage. The forest traps were set under trees and next to fallen logs. The field traps were set under thickets and in natural pathways.

On September 26th at 8:00, the traps were collected and trapped mammals examined. Data collected from caught mammals included: species name, sex, weight, age and reproductive condition. Age was categorized as juvenile, sub-adult and adult. Age was determined by the individual’s size and weight. For *P. leucopus*, pelage coloring was also used to categorize age. The juveniles have overall grey fur covering the back, the sub-adult have both brown and grey fur, and the adults have only brown fur. Reproductive condition for females was recorded as tiny nipples or enlarged nipples and obvious pregnancy was also noted. Reproductive condition for males was determined by the location of the testes, either undescended or descended. All captured mammals were then returned to their original transects.

**RESULTS**

Data from surveys conducted over the past 5 years was compiled and used to calculate overall trapping success of all species trapped. A periodical trend in population abundance can be noticed throughout the years for which data is available. The percent success of trapping for *P. leucopus* shows a decline from 18.0% in 2009 to 8.0% in 2010 (Fig. 1).
Figure 1. Percent trapping success of mammals surveys in the past five years in George Reserve.

Localized trapping efforts of both forest and field surveys show that the distribution of females and males of *P. leucopus* is very similar across both habitats. In both cases the amount of males in the area is slightly greater than the female population (Table 1).

Table 1. Trapping success of *P. leucopus* for 2010 survey. Percentage of the total number of individuals captured.

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Forest Abundance</th>
<th>%</th>
<th>Field Abundance</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Male</td>
<td>5</td>
<td>55.6</td>
<td>4</td>
<td>57.1</td>
</tr>
<tr>
<td>Female</td>
<td>4</td>
<td>44.4</td>
<td>3</td>
<td>42.9</td>
</tr>
</tbody>
</table>

The data obtained for the age distribution in the forest habitat show that the juvenile sector of the populations, in either of the sexes, is quite small: 20% males and 0% females (Table 2). The adult and sub-adult portions in males are much more abundant. In the case of females, adults are the only age group caught in the forest.

Table 2. Description of age for males and females in the forest. Percentage of the total number of individuals captured.

<table>
<thead>
<tr>
<th></th>
<th>Adult</th>
<th>Sub-adult</th>
<th>Juvenile</th>
</tr>
</thead>
<tbody>
<tr>
<td>Male</td>
<td>40.0</td>
<td>40.0</td>
<td>20.0</td>
</tr>
<tr>
<td>Female</td>
<td>100.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
</tbody>
</table>

In the field no juveniles, either male or female, were found. The distribution of adults and sub-adults is equal amongst females and skewed towards the sub-adult in males at 66.7%, while only 33.3% of the males found were adult (Table 3).

Table 3. Description of age for males and females in the field. Percentage based on individuals captured.

<table>
<thead>
<tr>
<th></th>
<th>Adult</th>
<th>Sub-adult</th>
<th>Juvenile</th>
</tr>
</thead>
<tbody>
<tr>
<td>Male</td>
<td>33.3</td>
<td>66.7</td>
<td>0.0</td>
</tr>
<tr>
<td>Female</td>
<td>50.0</td>
<td>50.0</td>
<td>0.0</td>
</tr>
</tbody>
</table>
Comparisons of reproductive conditions at the time of capture show that in the forest all males observed had undescended (abdominal) testes and that most females were either reproductively active or had recently been (Table 4). Only one fifth of the female individuals sampled were obviously pregnant, but 60.0% had enlarged nipples, possibly indicative of recent or ongoing lactation.

Table 4. Reproductive condition of males and females in the forest. Percentage based on individuals captured.

<table>
<thead>
<tr>
<th>%</th>
<th>Undescended Testes</th>
<th>Descended Testes</th>
<th>Pregnant</th>
<th>Enlarged Nipples</th>
<th>Tiny Nipples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Male</td>
<td>100.0</td>
<td>0.0</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Female</td>
<td></td>
<td></td>
<td>20.0</td>
<td>60.0</td>
<td>20.0</td>
</tr>
</tbody>
</table>

In the field, a more heterogeneous population was observed when comparing individual reproductive conditions. 25.0% of males had descended testes while 75.0% had undescended (scrotal) testes. Females had a proportion of 33.3% obviously pregnant individuals, with the remainder not currently showing ongoing or recent reproductive activity, as indicated by tiny nipples (Table 5).

Table 5. Reproductive condition of males and females in the field. Percentage based on individuals captured.

<table>
<thead>
<tr>
<th>%</th>
<th>Undescended Testes</th>
<th>Descended Testes</th>
<th>Pregnant</th>
<th>Enlarged Nipples</th>
<th>Tiny Nipples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Male</td>
<td>75.0</td>
<td>25.0</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Female</td>
<td></td>
<td></td>
<td>33.3</td>
<td>0.0</td>
<td>66.7</td>
</tr>
</tbody>
</table>

DISCUSSION

Based on the predictions of the source-sink model, we expected the healthier reproductive adults to inhabit the preferred environment (the ‘source’), which is the forest in this case. Non-reproductive juveniles and subadults would be outcompeted for habitat, and thus inhabit less preferred areas (‘sinks’), which is the field. We found that adult female *P. leucopus* were found in greater proportions in the forest than in the field, and no juveniles were found in the forest. Thus, the females followed what we would expect under the source-sink model, but the males did not.

We found that reproductive females most likely out-compete juvenile females for optimal habitat but males do not; this was expected as it has been reported in previous studies that females display a strong territorial behavior during the breeding season while males do not (Linzey and Brecht, 2005). This observation is supported by the fact that we found signals of active reproductive behavior in the trapped females, as enlarge nipples that indicate current lactation and a couple pregnant individuals, which indicate that they were actually in the middle of a breeding season at the time of the sampling.

*P. leucopus* males almost follow the reverse of the trend observed in females, and tend to be more reproductively active in the field. This may be likely due to the fact that they do not
provide any type of parental care of their offspring (Aguilar 2008), thus they do not have such a strong need to dominate resources during the breeding seasons as the females. Furthermore, adult males may encounter more mating opportunities with maturing females in the field, and after mating, the females move into the forest.

We also noted that the population abundance from this year follows the trend of a periodic rise and fall of population abundance in *P. leucopus* on the George Reserve. This trend follows a similar pattern from trapping efforts from previous surveys conducted at the University of Michigan’s Biological Station, in the lower northern peninsula of Michigan (Phil Meyers, pers. comm. 2010). The mechanism behind this fluctuation in population abundance is not known, but we would expect large *P. leucopus* populations (indicated by high trapping success) to follow mast years. With greater food availability, you expect greater survival and reproduction. Our results follow a cyclical pattern, with the last high trapping success occurring in 2006, dropping for a couple of years, back up in 2009, and slightly decreasing in 2010. This indicates that mast years likely occurred in 2005 and 2008, following by population booms in 2006 and 2009 respectively. Additional trapping surveys along with data on masting events in following years would give a better estimate of how closely the *P. leucopus* population follows the abundance of food sources.

ACKNOWLEDGEMENTS

We would like to thank Phil Myers for giving us the knowledge we needed to complete this project.

LITERATURE CITED


INFLUENCE OF CANOPY DIVERSITY ON DECOMPOSITION OF LEAF LITTER IN A TEMPERATE FOREST

RAFAEL D’ANDREA, CINDY BICK, MARK HUNTER, BINBIN LI, MINGQI LIU, KATE ZEMENICK

ABSTRACT

Studies of the impact of biodiversity on ecological function in “green” food webs are aplenty in the literature. The same is not true for decomposer food webs, and the effects of tree diversity on soil ecology are not well documented and still poorly understood. This study empirically establishes a negative correlation between canopy diversity and leaf litter mass in the E. S. George Reserve. Our results point toward an enhancing effect of diversity on decomposer function, a result in line with the general findings in terrestrial food webs.

INTRODUCTION

There have been many studies about biodiversity and ecosystem function in terrestrial food webs. Diversity conserved in agroecosystems has been shown to maintain fertile soil (Giller et al. 1997) and natural predators of pests (Vandermeer et al. 2002), thereby using diversity as a means for sustainable crop production.

Plant inter- and intraspecific diversity have been shown to stabilize grassland communities due to the asynchrony of species fluctuations in response to environmental variability (Hector et al. 2010). This study was in agreement with Tilman et al. (2006), who found that regardless of temporal scales, diversity in a grassland ecosystem was always associated with higher net primary productivity (Tilman et al. 2006).

However, convincing these investigations relating diversity to productivity and stability have been, some have attributed the real cause of productivity and stability to underlying factors, or “hidden treatments” (Huston, 1997). Huston (1997) introduced the idea that “hidden treatments” such as abiotic conditions, non-random selection of species whose effects overarch diversity alone, and the increased probability of including such species when diversity is increased, need be considered before attributing effects to diversity per se.

Additionally, Ball et al. (2009) brings attention to how additive and non-additive effects between species may alter the effects of diversity. Non-additive effects can be attributed to the interactions of certain species, which may only come about in a diverse system if the two species are present. Therefore, if non-additive effects are present, the identity of species present in a community cannot be ignored, or lumped into the category of diversity.
In terrestrial ecosystems, plant litter is the main resource of energy for soil microbes, and their subsequent breakdown of litter is a key ecosystem process for nutrient release (Hattenschwiler et al. 2005). However, there have been fewer studies addressing the relationship between biodiversity and ecosystem processes in “brown” or decomposer food webs. Of these recent studies, biodiversity has been found to reduce nitrogen and phosphorus release from litter (Ball et al. 2009) but also increase the diversity of soil biota (Ball et al. 2010). These studies show that litter diversity can have additive effects, which can be predicted by the effects of single species, or non-additive effects, which result from species interactions. Therefore, while litter diversity alone can beget diversity of soil microbes, different combinations of litter species interactions can alter nutrient release.

These recent studies have begun to identify the effects of litter diversity on nutrient dynamics and their non-linearities. All of these studies have been experimental in nature. The question therefore remains how plant diversity, by affecting soil ecology, impacts ecosystem function. This study seeks to understand if more diverse clumps of canopy trees in a temperate forest affect the decomposition of litter on the forest floor. Under the hypothesis that canopy diversity begets decomposer biodiversity, one predicts that diverse patches should accelerate decomposition, and thus litter amount should correlate negatively with canopy diversity.

MATERIALS AND METHODS

The experiment was carried out in the E.S. George Reserve, in September 2010. We set up 0.50m x 0.50m quadrats on the forest floor from which were collected all leaves which were not advanced enough in the decomposition process to have become detritus. Green leaves, twigs and living organisms were not included in our measurements. Around each quadrat, a 5m-radius circle was drawn and every tree falling within this 78.5m² patch of the woods whose girth at breast height (GBH) was >10cm was identified and had its GBH noted down. We collected leaves from twenty quadrats, and strived to cover a good range of diversity between patches, from monoculture to the most diverse spots we could find. The data was all gathered in one morning.

Bag-loads of leaf litter were then weighed with a digital scale, and the litter was sorted in two groups: oak leaves and leaves from other genera. Species abundance was calculated in terms of DBH (DBH = GBH/3.14), not individuals, and the Shannon diversity index was used. DBH is used in this study as proxy for the amount of leaves in the surrounding canopy.

RESULTS

Fig 1 shows the DBH composition of each of the twenty plots, in increasing order of DBH diversity.
In order to single out the role of diversity per se from the influence of other factors such as leaf amount in the canopy, dominant species (species with disproportional or peculiar effects), and non-additive effects brought about by species interactions (Huston 1997), we performed multivariate linear regression analysis. Several models were compared to determine which set of the abovementioned items best predicted the leaf litter amount. The results are summarized in Table 1.

Table 1. Results of various linear regression models. totlit: total quadrat litter; totdbh: total quadrat DBH; oaklit: total oak litter per quadrat; noaklit: total litter from other genera per quadrat; noakdbh: total DBH from other genera per quadrat; div: Shannon index. AIC: Akaike Information Criterion, informs model quality. Smaller AIC indicates a better model.\(^1\)

| Model                      | AIC   | \(R^2\) | DBH | \(Pr(>|t|)\) | DIV | \(Pr(>|t|)\) | Oak | \(Pr(>|t|)\) |
|----------------------------|-------|---------|-----|--------------|-----|--------------|-----|------------|
| totlit \(\sim\) div       | 196.7 | 0.27    | --  | --           | -103.5 | 0.02    | --  | --         |
| totlit \(\sim\) totdbh    | 202.4 | 0.02    | 0.10| 0.5          | --   | --           | --  | --         |
| oaklit \(\sim\) noakdbh   | 198.3 | 0.48    | 0.55| 0.007        | --   | --           | --  | --         |
| oaklit \(\sim\) noakdbh + div | 200.3 | 0.48    | 0.55| 0.001        | 7.7  | 0.86        | --  | --         |
| noaklit \(\sim\) noakdbh  | 195.9 | 0.33    | 0.32| 0.008        | --   | --           | --  | --         |
| noaklit \(\sim\) div      | 198.9 | 0.23    | --  | --           | -98.7 | 0.03     | --  | --         |
| noaklit \(\sim\) noakdbh + div | 189.6 | 0.56    | 0.32| 0.002        | -98.1 | 0.009    | --  | --         |
| noaklit \(\sim\) noakdbh + div + oakdbh | 180.5 | 0.75    | 0.17| 0.05         | -106.1 | 0.0009 | -0.34 | 0.003  |

\(^1\) Let \(n\) be the number of observations and \(k\) be the number of parameters in a given model. \(AIC = n \ln[(1-R^2)/n] + 2k.\)
Regression of total quadrat litter against diversity yields significant correlation, indicating that diversity indeed impacts the decomposition of leaf litter (Fig 2). However, total quadrat DBH surprisingly did not correlate significantly with litter. As this could indicate the presence of a dominant species, we sorted oak litter (the most common genus) from the rest and ran the simple “litter ~ DBH” test for both resulting groups. This time correlations are observed.

Whereas oak litter does not seem to be affected by diversity, the anticipated correlation is in fact observed in the case of litter from other species. The best model we obtained, explaining 75% of the variance and showing the smallest AIC index, was

\[
\text{non-oak litter} \sim \text{non-oak DBH + diversity + oak (dominant species)}
\]

Further introduction of predictors such as other dominant species candidates and interaction terms produce less significant models, indicating that we correctly identified the best predictors among the data we collected.

**DISCUSSION**

The expected impact of canopy diversity on litter decomposition can be said to have been observed in this experiment. We seem to have observed an “identity effect” from a dominant genus, *Quercus*, which seems to affect the amount of leaf litter of other genera occurring in the George Reserve but whose litter is not itself affected by their presence. This might be due to tannins released by oak leaf litter, which have been shown to inhibit the activity of at least four important fungal decomposers (Harrison 1971). Our results, therefore, support the claim that identity effects on litter decomposition might be an
important force (Ball et al. 2009). On hindsight, this is but expected, as species identity is certainly an important factor on herbivory (herbivores show clear preference and specificity towards specific plants, presumably partly due to differentiated digestibility).

A corollary of this is that, even though total litter correlates with diversity, such correlation appears to be effected through the “non-oak” genera. Therefore, it seems more meaningful to run regressions on these two groups separately, as done in this study.

A few possible reasons for this negative correlation between litter and canopy diversity can be cited. Presumably, such diversity brings about an assortment of decomposing organisms, each of which specializing for their own set of compounds but, when brought together, would be able to take apart the leaf matter more efficiently and thoroughly. Though certainly plausible and even commonly assumed (Ball et al. 2010), this idea needs further testing. Another possibility is that the mix of nutrients originating from the variegated leaf stock accelerates decomposition rates (Liu et al. 2009, Manzoni et al. 2010).

The lack of significance between total quadrat litter and total quadrat DBH may be in part due to the clearly less-than-adequate radius size of the area encircling the quadrats. During data collection, we observed many trees lying outside and therefore could not be included that clearly contributed to the litter being gathered. We even observed 5m-radius patches devoid of trees which nonetheless contained leaf litter. This is likely to introduce a not-to-be-overlooked sampling effect, to be handled in future runs of this experiment.

It should also be interesting to repeat this experiment on other ecosystems to verify how general this influence of biodiversity on decomposition function is.

We would like to open a parenthesis here to comment on the perhaps surprising higher linear correlation of litter with DBH than with \((DBH)^3\). As mentioned above, DBH was used here to stand for the amount of leaves in the canopy, which obviously correlates linearly with litter. Assuming that the amount of leaves a tree produces is linear on its volume, one concludes that litter should be linear on a cubic quantity, such as \((DBH)^3\), instead of on a linear quantity like DBH. A possible explanation, assuming the \((\text{volume} \rightarrow \text{No. of leaves})\) relationship to hold, is that trees do not keep their proportions constant (they might keep their height relatively constant while their girth changes significantly or vice-versa) during some or most of its life. Another possibility is that the number of leaves correlates better with metabolism than plainly with biomass.

ACKNOWLEDGEMENTS

We would like to thank Prof Mark Hunter for helping with the conception and execution of the project.
LITERATURE CITED


ABSTRACT
We studied competition and dominance in ant species. Our focus was on the three most populous ant species in the field near the experimental ponds in the George Reserve: *Solenopsis molesta*, *Monomorium minimum* and the invasive *Tetramorium caespitum*. We tested the foraging capabilities of the ants by baiting with tuna fish, a highly nutritious food source. Repeated sampling of the study plot allowed us to estimate species dominance at each bait location. Competition coefficients were measured as a function of species dominance over the food source. We found that these three major ant species form an intransitive loop whereby each species is both dominant over one species and dominated by the other.

INTRODUCTION
Mathematical modeling has become one of the fundamental tools to predict the future of many organismal communities. Ecological models attempt to encompass and represent the main mechanisms that drive detrimental population dynamics. One of the canonical examples of an ecological model is the Lotka-Volterra model, which seeks to provide a framework for understanding competition amongst species. This competition model takes into consideration various variables that determine overall survivorship of competing species. These variables tend to approximate the outcome of a population based on the density, carrying capacity, and growth rate of a given community. Although the model is very strong it does not consider the effect of spatial dynamics when referencing the competitive interactions of two or more species. One of the shifts in ecological theory is the need to consider and address the intrinsic importance and effect that spatial arrangement has on the success of various systems.

The main driver of various competition models is the competition co-efficient, which quantifies the effect of a species upon another when referencing the ability of preemption of various necessities, in comparison to a closely neighboring species. This co-efficient, however, doesn’t necessarily encompass the direct effect of species A upon species B by restricting it from all the resources that it needs. The co-efficient may also represent the strength by which a particular species is able to deflect the effects of a strong ‘effect’ competitor, thus making it a strong ‘response’ competitor (Goldberg & Fleetwood, 1987). Yet it has been observed that these competition co-efficients can only predict so much about the dynamics of a particular system. The spatial components have unfortunately seldom been considered in classical ecological theory. These special variables add a whole new level of complexity when examining the dynamics of various species co-existing with one another.

An optimal model system that shows the importance of space in reference to the outcome of community structure and survivorship are ant communities. Ants have been known to form intricate and dynamic mosaics that are comprised of various competing species of ants living in close proximity to one another (Blünthgen & Stork, 2007). The formation of these seems to allow various species, both strong ‘effect’ competitors and non, to coexist in the presence of one
another (Yitbarek & Vandermeer, unpublished 2010). In a study currently being conducted by Yitbarek and Vandermeer, it has been noted that these patterns are a compilation of what they refer to as ‘mosaic species’ and ‘satellite species’ maintaining a dynamic balance. The mechanism behind this balance is a series of competitive mechanisms tugging on one another in what can be thought of as an intransitive loop (Yitbarek & Vandermeer, unpublished 2010). Various studies have concluded that these loops tend to increase biodiversity, however, recently Vandermeer (2010) has argued that rather than increase diversity these loops tend to suppress diversity. The species that form these loops keep each other in check in terms of their competitive strategies by harboring species that may not survive on their own ‘satellite species’ without the protection of ‘mosaic species;’ yet these loops still tend to drive out other surrounding species that have to compete with these mosaic formations (Vandermeer 2010).

The following study was conducted at the E.S. George reserve in a uniform field using baits to determine the competitive strategy and overall territoriality of three common to the area ant species. This project also looks at space as a determinant of the biodiversity in a given area predetermined by the competitive strategies of (Solenopsis, Tetramorium, and Monomorium). We seek to observe the community and spatial structures of the ant communities surveyed and thus be able to calculate competition co-efficients to use in a model recently developed by Yitbarek and Vandermeer (2010). We also aim to understand, through the output of this model, the underlying mechanisms that could possibly produce coexistence or ousting of ant species from a particular area.

**METHODS**

**Study Site**

The study site was a fairly homogeneous 90x30 meters section of land located at the ESGR in E.S. George reserve. This relatively flat area was created by heavy machinery several years ago for the construction of the experimental ponds. It is characterized by as a grassland cover and it has a rich insect community.

**Data Collection**

This study focused on the ant species that have been reported as most common in the area during the past few years: Solenopsis molesta, Monomorium minimum and Tetramorium caespitum (Iverson et. al 2009). S. molesta and M. minima are a native species to the United States while T. caespitum is an invasive species introduced from Europe. Ant baiting was performed on the September 25th 2010 at 8:30am. Temperature was 56 degrees Fahrenheit. Tuna baits were placed every 2 meters in 14 transects distributed in a grid-like structure. After 30 minutes the baits were examined and the present species were recorded. The baits were examined again 4 hours later, species recorded with indication of dominant species. After 24 hours we checked the baits again and only recorded the presence of S. molesta, M. minima, and T. caespitum to determine the competition interactions of the species.

**Data Analysis**

Competition coefficients were calculated based on the presence or absence of the ant species on each bait in each round. Competition coefficients were used to run a species interaction model to try to understand the spatial patterns of the mosaic.
RESULTS

The presence of each ant species at each of the 644 bait sites set on a 14 X 46 grid were recorded at three sampling times: thirty minutes after bait placement, four hours after placement, and twenty-eight hours after placement. The presence of the three major species (Solenopsis molesta, Monomorium minimum, Tetramorium caespitum) were plotted on a scatter plot representing the survey area, with the higher numbered rows corresponding to the eastern end of the plot and lower corresponding to the western end (Fig. 1).

Figure 1. Graphical representations of the presence of the three major species in each survey

This graphical representation of the entire surveyed plot shows the increase in total ants present from the initial survey to the third survey. A pattern of species distribution can also be seen in the second survey plot, with the western (lower in Fig. 1) and central sites mostly occupied by S. molesta, the eastern (upper) sites occupied by T. caespitum, and M. minimum distributed throughout the plot. The percent presence of each of the three species (of all sites occupied) was calculated for each survey (Fig. 2). Here it is shown that S. molesta maintained a similar percentage of all occupied sites, M. minimum increased in percentage of all occupied sites, and T. caespitum reduced in percentage of all occupied sites over the surveyed time period.
The number of shared bait sites and which species were occupying those sites was also calculated from the third survey data. All three major species shared 7% of all sites (46 sites), *S. molesta* and *M. minimum* shared 21% of all sites (133 sites), *M. minimum* and *T. caespitum* shared 10% of all sites (65 sites) and *S. molesta* and *T. caespitum* shared 2% of all sites (12 sites). Competition coefficients between each of the three major species were calculated by taking the number of site takeovers by one species in the second survey from the other species in the first survey and dividing by the total number of nests occupied by the species taken over in the first survey (Fig. 3). The resulting coefficients show that *S. molesta* usually outcompetes *M. minimum*, *M. minimum* usually outcompetes *T. caespitum*, and *T. caespitum* usually outcompetes *S. molesta* (Fig. 4).

![Figure 2. Percent presence of each species in all occupied bait sites](image)

<table>
<thead>
<tr>
<th>Species</th>
<th>Survey 1</th>
<th>Survey 2</th>
<th>Survey 3</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Solenopsis molesta</em></td>
<td>41%</td>
<td>43%</td>
<td>42%</td>
</tr>
<tr>
<td><em>Monomorium minimum</em></td>
<td>19%</td>
<td>22%</td>
<td>29%</td>
</tr>
<tr>
<td><em>Tetramorium caespitum</em></td>
<td>40%</td>
<td>35%</td>
<td>29%</td>
</tr>
</tbody>
</table>

Figure 3. Equation used to calculate the competition coefficients between the three major species

\[ \alpha_{(1,2)} = \frac{\# \text{ of sites with sp. 2 in survey 2 and sp. 1 in survey 1} - \# \text{ of sites with sp. 2 and sp. 1 in survey 1}}{\text{total \# of sites with sp. 1 in survey}} \]

Figure 4. The calculated competition coefficients between each pair of the three major ant species, where \( \alpha_{(1,2)} = \text{likelihood of species 2 overtaking species 1} \). S= *S. molesta*, M= *M. minimum*, T= *T. caespitum*

<table>
<thead>
<tr>
<th></th>
<th>( \alpha_{(S,M)} = 0.19 )</th>
<th>( \alpha_{(S,T)} = 0.13 )</th>
</tr>
</thead>
<tbody>
<tr>
<td>( \alpha_{(M,S)} = 0.52 )</td>
<td>( \alpha_{(M,T)} = 0.08 )</td>
<td>( \alpha_{(T,S)} = 0.07 )</td>
</tr>
<tr>
<td>( \alpha_{(T,M)} = 0.16 )</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
DISCUSSION

Up to and including this third year of study, *T. caespitum*, *M. minimum*, and *S. molesta* coexist without a clear indication of competitive exclusion altering levels of co-dominance (Jackson et al. 2008; Brady et al. 2009). This is discordant with traditional niche theory (Hutchinson 1957) which argues that no two species can occupy the same niche for a prolonged period of time. Given that our study site was uniformly scraped and cleared in 1988, and still represents relatively homogeneous grassland in the initial stages of forest successsion, one would predict a specialized ant competitor to dominate the given tuna baits in time or space. Yet this is not the case (Fig. 1, Fig 2). Here we investigate the mathematical concept of intransitivity as a potential mechanism for coexistence in the grassland stages of an early-successional temperate forest.

In ecological theory, an intransitive loop occurs when species A > species B > species C >species A, etc. Intransitive loops result in a constant rotation of species distributions, wherein each species is driving out an inferior competitor as it in turn is being driven out by a superior competitor. Our competition coefficients support our hypothesis that an intransitive loop is underlying species coexistence of *T. caespitum*, *M. minimum*, and *S. molesta*. Specifically, *S. molesta* out-competes *M. minimum*, which out-competes *T. caespitum*, which out-competes *S. molesta*. Yet in the case of *M. minimum* and *T. caespitum* overall competition is weak, and the directionality of competition is marginal (e.g. alpha TS 0.07; alpha ST 0.13). Indeed, our competition coefficients were unable to produce a perpetuating intransitive loop in our computer simulations; *S. molesta* consistently eradicated all competitors. Nonetheless, in conjunction with steady emigration from nearby forest habitats, the circular nature of inter-competition between *T. caespitum*, *M. minimum*, and *S. molesta* may contribute to their continued coexistence.

*Monomorium minimum* as a mediator of diversity

*M. minimum* was present at over 95% of co-shared bait sites in our final sample, while *S. molesta* and *T. caespitum* only co-shared 5%. This curious level of presence may indicate that *M. minimum* is not a good competitor *per se*, but good at pacifying competition. Anecdotally, we observed that *M. minimum* would halt and extend its abdomen into the air after provocation by another species (usually *T. caespitum*). The offending species would then avoid the *M. minimum*. Later, we found that after provocation with a stick, *M. minimum* would likewise extend its abdomen, again presumably emitting a defense pheromone. If this defense compound disarms competitors, *M. minimum* may be acting as a mediator between species, allowing co-existence. This hypothesis is supported by the difference in co-shared sites by *T. caespitum* and *S. molesta* (5%) and sites shared by all three species (18%).
WORKS CITED


VARIANCE OF FUNGAL DIVERSITY AND ABUNDANCE ALONG PLANT COMMUNITY, SUBSTRATE, AND FUNCTIONAL LINES

IMAN SYLVAIN, SUSAN CHENG, DAVID GONTHIER, ABIGAIL HYDUKE, AND WILLIAM WEBB

ABSTRACT

Fungi are a widely dispersed and highly diverse group of organisms. Fungi are integral to the functioning of many ecosystems because they play a vital role in nutrient cycling. Fungi are especially important to plant communities, not only as decomposers of plant material, but also in their nearly ubiquitous mutualistic associations with plant roots as mycorrhizae. Despite the essentiality of fungi in ecosystems, little is known about fungal diversity in distinct ecosystems. This field study investigated whether difference in fungal diversity and abundance are correlated with differences in plant community structure, and whether fungal species or functional groups segregate among substrates. Three fungal surveys were conducted along 25-meter transects in a red pine conifer forest, deciduous forest, and an old field. Collected sporocarps (fungal fruiting bodies) were identified to species, and their relative abundance, host substrate, and functional classification were recorded. Results of this study show that fungal communities differ in abundance and diversity along forest types and substrate lines. The deciduous forest held the greatest abundance and species diversity of sporocarps, followed by the pine forest, and lastly the old field. The primary functional group found within all forest types were decomposers.

INTRODUCTION

Fungi provide essential services to plant ecosystems and play a pivotal role in ensuring proper ecosystem functioning. Certain fungi, such as saprotrophs, allow plant material to continue cycling in an ecosystem by breaking down dead plant tissue (Newbound et al. 2010). Many fungi convert energy and nutrients, such as nitrogen and phosphorus, into chemical forms that would otherwise be unavailable for plant use (Helgason et al. 2002, Newbound et al. 2010). By providing plants with the nutrients they need to photosynthesize and store carbon, fungi indirectly influence the carbon cycle (McGuire et al. 2010, Treseder and Cross 200). In addition to cycling nutrients, fungi influence soil structure by aggregating soil together with their hyphae (Newbound et al. 2010).

Fungi are divided into five groups, the Chytridiomycota, Zygomycota, Ascomycota, Basidiomycota, and Glomeromycota (Newbound et al. 2010) and are comprised of three functional types, mutualists, decomposers, and parasites. Approximately 95% of terrestrial plants form mutualistic mycorrhizal relationships with fungi. Specifically, 60% of plants develop
interactions with arbuscular mycorrhizal fungi (Newbound et al. 2010, van der Heijden et al. 2003). The arbuscular mycorrhizal fungi are part of the Glomeromycota and convert nutrients into available forms for plants while receiving carbohydrates from their hosts (Newbound et al. 2010, Treseder and Cross 2006, Vellinga et al. 2009). The Ascomycota and Basidiomycota groups include ectomycorrhizal fungi (Vellinga et al. 2009), which form similar mutualistic partnerships with plants.

Although fungi are essential to ecosystem functioning, much is still unknown about fungal diversity, especially in certain forest types (Newbound et al. 2010, Opik et al. 2008). One goal of this study was to determine whether plant communities (conifer forest, deciduous forest, and an old field) are correlated with fungal diversity, abundance, and functional groups. Another goal was to determine whether fungi segregated according to substrate. We predicted that the highest diversity and abundance of fungi would be found in the deciduous forest due to its heterogeneity of substrates and reduced proportion of ectomycorrhizal fungi, which may suppress decomposers. We also expected that the lowest diversity would be found in the old field due to a lack of woody substrates in that plant community. Since previous research has shown that fungi in forests can be functionally diverse in substrate preference, we predicted that fungi would separate according to substrate (McGuire et al. 2010).

**METHODS**

This study was conducted at the E.S. George Reserve in Livingston County, Michigan on September 25, 2010. Fungal surveys were conducted in three distinct habitats to investigate whether fungal diversity was specifically correlated with different ecosystems. The three study sites included an even-aged red pine (*Pinus resinosa*) plantation, a mixed deciduous forest, and an abandoned agricultural field dominated by *Eleagnus umbellata* and *Juniperus virginiana*. In each forest type, three transects of 25 meters each were set up by following the direction of a randomly dropped pen. Four designated surveyors collected all visible sporocarps within a meter of the transect line. Fungal fruiting bodies were preserved in plastic bags and transported back to the lab. The abundance of the fungi in each transect, as well as its host substrate were recorded. Using North American Mushroom: A Field Guide to Edible and Inedible Mushrooms, sporocarps were identified to the species level whenever possible, and to genus if not. Specimens that were unidentifiable were deemed ‘unknown’. The phylum and functional group of specimens were also noted.

**Analysis**

To compare the use of substrate across different habitat types, the number of individuals and the total number of species found on each substrate type was compared with Chi-Squared tests. To compare the different functional types across different habitats, the number of individuals and the number of species within each functional group was compared with Chi-Squared tests. To examine species diversity and abundance across the different habitats, species accumulation curves were created in EstimateS 820. The estimated mean number of species per sample was plotted using a line graph with respective 95% confidence interval. To statistically determine differences between curves, the overlap between confidence intervals was measured on graphs. We then used the program PAST for nonmetric multidimensional scaling (NMDS) and analysis of similarities (ANOSIM) to statistically compare species composition of fungi across habitats. In both the NMDS and ANOSIM analyses, we considered each transect a
replicate and used the Bray-Curtis similarity index. The ANOSIM produces a global p value that indicates differences in species composition and reports pairwise comparisons between particular sites.

RESULTS

Figure 1. This graph shows the number of fungal species found on each substrate type in the deciduous forest.

![N Species Found in Deciduous Forest Substrate Types](image)

We discovered there was a significant relationship between the substrate type and the level of biodiversity found in the deciduous forest community (Figure 1). A Chi-Square test revealed a significantly higher number of species found in soil substrates (P < 0.001) than the four other substrate types based on the number of individuals expected. No other forest community had an observed correlation between substrate type and its level of biodiversity.

Figure 2. This graph shows the number of individual fungi found on each substrate type in the conifer forest.

![N Individuals Found in Conifer Forest Substrate Types](image)

The total number of individuals found on log substrates in the conifer forests was significantly higher (P < 0.001) than the number of individuals found in the other substrate types.
We found a similar relationship in log substrates of the deciduous forest community, although it was not as strong (P < 0.01). Additionally, the deciduous forest community has a higher number than expected of individuals found in soil substrates (P < 0.01) and a weaker result (P < 0.05) in the mixed soil and leaf litter substrate (Figure 3). There are much fewer (P < 0.001) individuals found in leaf litter substrate of the deciduous forest (Chi-Squared).

Figure 3. This graph shows the number of individual fungi found in each substrate type in the deciduous forest.

There was a higher number of individuals observed than expected found in the branch substrate type of the old field forest community (Figure 4). This correlation had a significant value (P < 0.01) determined by a Chi-Square test. The old field was the only forest community that had a significantly larger number of individuals found on the branch substrate.
Figure 5. This graph shows the number of species within a functional group found in the deciduous forest.

![Deciduous Forest N Species of Functional Types](image)

Figures 5 and 6 illustrate the significantly higher proportion of decomposer species found in our study. Of the three functional groups identified and the added possibility of not being able to determine a species’ functional types, we were still able to detect higher than expected observations of decomposers (Chi-Squared Test) (Deciduous $P=0.01$) (Old Field $P=0.05$).

Figure 6. This graph shows the number of species within a functional group found in the old field.

![Old Field N Species of Functional Types](image)
Figure 7. This graph shows the number of individuals within a functional group found in the conifer forest.

![Conifer Forest N Individuals of Functional Types](chart)

The total number of individuals from the decomposer functional type was significantly higher than the other functional types observed (Figures 7 and 8). In both conifer and deciduous forests, the Chi-Square test confirmed that the number of observed individuals whose species can be described as decomposer was much higher than expected (P = 0.001). This relationship was compounded with the lower than expected observations of parasitic functional groups in both forest communities, as well as the lower than expected number of ectomychorrhizal individuals collected from the deciduous forest.

Figure 8. This graph shows the number of individuals within a functional group found in the deciduous forest.

![Deciduous Forest N Individuals of Functional Types](chart)

Species Diversity

The relationship between species accumulation per sample attempt (transect) shows that none of the three habitats sampled begin to asymptote after three transects (Fig. 9). This suggests that the sample transects did not capture the entire community of species. However, even with
this small sample size, the 95% confidence intervals do not overlap between communities, implying that the deciduous community had more species than the conifer forest and old field, and the conifer forest also had more species than the old field (Fig. 9).

Figure 9. This graph shows a Species Accumulation Curve across transect samples in the deciduous forest, conifer plantations, and old field.

Figure 10. This graph shows a Non-metric Multidimensional Scaling of the relative abundance of species across the deciduous forest, conifer plantation, and old field.
Species Composition

The NMDS plot and ANOSIM analysis suggests fungal species composition tended to differ between the three habitats (global $R = 0.65$, $P = 0.0096$). Fungal communities in the deciduous forest were marginally different from the conifer ($P = 0.096$) and old field ($P = 0.096$) communities. However, old field and conifer communities were similar ($P = 0.31$) (Fig. 10).

DISCUSSION

Our survey of three different forest communities revealed that the abundance and diversity of fungal populations was directly correlated to the composition of the forests they inhabited. Each of the forest communities had different attributes that we hypothesized would be reflected in our fungal survey data. We anticipated that the contrasting life histories of the coniferous, deciduous, and old field would yield very different substrate qualities. This may be the result of contributed nutrients, or lack thereof, by the forest populations. The nutrient quantities supplied by abscission (the dropping of the leaves in autumn) of deciduous trees, and the accumulation of down woody debris in the deciduous forest seems to be unmatched in the coniferous forest. Coniferous forests typically exist in poorer soils with high acidity and low nutrient levels. The presence of conifers in these environments continues to degrade the soil as acidic pine needles are shed. As a result, conifer communities become very characteristic. Organisms that are able to live in conifer forests have tolerances similar to those of conifers or adaptive strategies that enable to them to survive in the harsh environment. The distinct characters of our two forest types and the relative age of the disturbed forest were the foundation of our hypotheses and support the implications made in our data analysis.

Species Diversity and Composition

Species diversity differed across habitats but our sampling method did not exhaust all species within the community. The deciduous forest may have had higher species diversity for several reasons. First, more productive systems like deciduous forests and temperate grasslands tend to have high soil species diversity (Allen et al. 1995, Bardgett 2005). It is also likely that deciduous forests contain high quality mull humus types, (organic particles which no longer break down) that support high microbial biomass and are typically correlated with great microbial species and functional diversity (Ponge 2003). The conifer plantation and old field are likely to be moder or mor humus types, which may support less biological activity. These humus types are generally less heterogeneous than mull, and therefore support lower microbial biodiversity (Ponge 2003).

Deciduous forests are also patchier than conifer plantations or old fields. Patches of trees dominated by ectomycorrhizal fungi and trees dominated by arbuscular mycorrhizal fungi may coexist in a forest. The combination of these two patches might substantially increase fungal diversity because it allows for mycorrhizal communities to differentiate. This patchwork also supports decomposer fungi communities which may compete with ectomycorrhizae for resources. At local scales this level of patchiness can result in great differences in fungal diversity (Bardgett 2003). Greater diversity of deciduous and conifer forests, relative to old field, may have been due to differences in substrates available to the fungal community. In forest habitats there are numerous substrate resources for fungal decomposers and mycorrhizae to
utilize. Differences in substrate availability are likely to result in differences of species diversity (Tedersoo et al. 2003, Bardgett 2005).

The observed species composition was marginally different between the deciduous forests compared to the conifer forest and old field, but not between the old field and conifer forests. The fact that there were only marginally statistical differences may be a result of our low sample size. Further sampling would likely establish stronger statistical relationships. Nonetheless, differences in species composition may have resulted from differences in dominant plant species of each community. Differences between communities probably resulted from similar processes that constructed the species diversity. These factors being mainly the patchiness of the habitat, humus types, difference in floral community, and differences in substrate availability (Ponge 2003, Tedersoo et al. 2003, Bardgett 2005).

**Deciduous Forest Community**

The deciduous forest had higher levels of fungal abundance and biodiversity than any of the other forest communities we surveyed (Figure 10). This supported our prediction of deciduous forests providing higher quality habitat for fungi due to enhanced soil conditions. Increased nutrient availability in the soil and organic matter (dead leaves, branches, or logs) on the forest floor may provide optimal conditions for fungal populations to thrive. This is emphasized by our data. We observed significantly higher biodiversity of fungi found in the soil substrates of our survey (Figure 1). The significantly higher abundance of fungi observed on logs and soil suggest that the deciduous forest would be suitable habitat to individuals not only absorbing the nutrients directly from the ground (soils substrate), but also aiding in the decomposition of the logs and leaf litter. This assumption is validated by the significantly higher number of species which are classified functionally as decomposers (Figures 5 and 8).

**Coniferous Forest Community**

We would expect to see less fungal biodiversity in a conifer forest if there was indeed a difference in the soil quality. Low fungal biodiversity would theoretically reflect poorer soil conditions. This is exactly what we observed. Fungal abundance was highest on the log substrates (Figure 2). This corresponds with our predictions because individuals may have to derive their nutrients directly from logs if they are not able to absorb enough nutrients from the soil. This might also mean that the conifer forest would host a higher number of decomposer fungi, which we also observed (Figure 7). As predicted by our species accumulation curve after resampling our data, there is a significantly higher number of species and individuals observed in the deciduous forest than the conifer forest. Under the assumption that the composition of the deciduous forest is similarly aged, the consistency of our data with the expected results suggests that deciduous forests would be the optimal location for long-term fungal community growth and expansion. In contrast, we would expect there to be less individuals present in a disturbed forest with younger growth.

**Disturbed Forest Community (Old Field)**

Our predictions for the disturbed forest community were found to be true. There were a limited number of individuals found throughout the old field plots. The most individuals were
found on the smaller branch substrates (Figure 4). This may suggest there has not been enough time for adequate organic matter to accumulate in the soils or large enough fragments of old growth to fall to the ground and potentially serve as log substrate. The significantly higher number of decomposers found in this site (Figure 8) suggests that the fungal community in disturbed forest has to be opportunistic. These fungi may possess the ability to capitalize on a variety of nutrient sources or risk high mortality rates.

Limitations of our Study

Although our data consistently reflected our predictions, inevitably there were factors that may have decreased the accuracy of this study. All of the fungi we surveyed belonged to the phylum Basidiomycota. Ultimately this may reflect the seasonality of this survey. Sporocarps in other phyla may have senesced before our collection. If this was the case, we acknowledge that ectomycorrhizal fungi may not have been fully accounted for. For future investigations it may prove beneficial for surveys to be conducted in the spring or summer in order to compare the diversity and relative abundances of fungi throughout the season. It may also be interesting to see if the populations correlate specifically with time of year collected, and what role that may have played in influencing our data.

Conclusions

Fungal diversity varies greatly across global and local scales. However there are few studies documenting these changes, and even fewer studies investigating why the changes occur (Bardgett 2005). Specifically, human alteration to landscapes can result in drastic reductions in fungal and microbial diversity (Bardgett 2005). Here data was presented that demonstrates how species diversity and composition of fungal communities was reduced or altered across three habitats. Stark differences in fungal diversity, individual abundance, and substrate preferences were shown in a native deciduous forest, conifer plantation, and an abandoned agricultural field.

ACKNOWLEDGEMENTS

We would like to thank Krista McGuire for leading us in this field problem and helping to identify all of our fungal specimens. Without her knowledge and assistance this project would not have been possible. We would also like to acknowledge our professors, John Vandermeer, and Ivette Perfecto, and to thank our Graduate Student Instructor David Allen.

LITERATURE CITED


APPLICATION OF ANT SPATIAL DYNAMICS TO MODELING SCALAR PATTERNS OF COMPETITION

WILLIAM WEBB, SUSAN CHENG, DAVID GONTHIER, ABIGAIL HYDUKE, IMAN SYLVAIN, JOHN VANDERMEER

Abstract. Differences in foraging strategies and the resulting spatial distribution of ants can yield stable mosaic patterns within a population. The interactions of three ant species Monomorium minimum, Solenopsis molesta and Tetramorium caespitum were used to quantify competition coefficients of the response to and maintained occupation of food resources on five 91cm by 91cm plots in a homogenous environment. These results were modeled in a 10 by 10 lattice to determine if fractals did exist when examining competition simulations. Our field data reflected our simulation results that at a 10 by 10 cell lattice cannot support multiple species and that one dominant ant species will emerge. We calculated a minimum threshold of a 50 by 50 cell lattice parameters as necessary in order to maintain each species of a population within a simulation for any substantial unit of time (at least 200 iterations). These data indicate that in general the stability of mosaic patterns in any competitive system is not uniform in conformation at different scales and that spatial constraints must be set above some minimum area to sustain biodiversity.

INTRODUCTION

Competition is a key factor to determining the abundance and distribution of species within a given community (Connell 1961). Competition has been fundamental to our understanding of coexistence. Further, as more and more complex systems begin to be studied, more complex patterns of competition are discovered. Intransitive loops are cases in which three or more species interact competitively so that species one beats species two and species two beats species three and species three beats species one; like a game of rock, scissors, and paper (Grace et al. 1993). Discrete competition models show that these particular three way competition interactions can be stable and maintain coexistence. They also can produce interesting spatial patterns when incorporated into spatial models. However, little is still understood about the effect of scale on these patterns and whether similar patterns of coexistence occur at smaller scales. In other words, do these intransitive loops act as fractals at lower scales?

Ant assemblies form a model system to investigate the importance and existence of intransitive loops within in natural ecological communities. Competition is a foundation in the ecology of ants (Hölldobler & Wilson 1990). Ant species often form mosaics and spatial patterns where dominant species exclude other sub-dominants resulting in a patchwork of species. The dynamics of these patches has been described as a competition-colonization trade-off (Levins & Culver 1971, Yu & Wilson 2001). A given species can be successful by having high colonization rates to new patches or by being strong competitors for space with existing species.

The purpose of this study was into investigate the interactions between three ant
species at a very small local scale using a discrete spatial model. Maloney et al. (2010) found evidence of an intransitive competitive loop between three ant species in the field by the experiment ponds at the George Reserve. Here we examine how this intransitive loop, and the spatial patterns it creates, operate at a small spatial scale.

METHODS

Study site

The study was conducted in the E.S. George Reserve in Livingston County, Michigan in a field where the topsoil was removed in 1988 to create a homogenous environment. This field rests next to the Evans experimental ponds. A permanent grid divides the space of 92m by 22m into 2m by 2 m plots.

Field experiment

Study sites were selected within the grid according to the presence of the three species of concern (*Monomorium minimum*, *Solenopsis molesta* and *Tetramorium caespitum*) in the previous day’s survey, which occurred on a larger scale.

Prior to establishing the plots, the vegetation was cleared to ground level to facilitate ant visibility. Five plots of 91 by 91 cm were established within the original grid. Within the plots tuna-baits were place every 10 cms for a total of 81 samples per plot.

Foraging ants of the aforementioned species were recorded at each bait on three separate occasions. The first survey took place at 11:18am, followed by 1:36pm and 3:30pm. Temperatures stayed fairly constant throughout the day 12-13 °C, but the first survey was conducted under clear conditions and the final two under heavy cloud cover.

Mathematical Model

Our model was constructed to simulate competition using an equation deriving discovery and dominance coefficients from foraging behaviors observed on the same homogenous environment 24 hours prior to our field study taking place. Using the data we collected during the field study, we modified the competition coefficients slightly by adjusting one of the coefficients in order to standardize the direction in which the competition coefficients followed, as this coefficient did not retain the intransitive loop formation that our earlier data was able to demonstrate. The resulting competition coefficients were arranged in an intransitive loop, with *Monomorium minimum* dominant over *Tetramorium caespitum*, *Tetramorium caespitum* dominant over *Solenopsis molesta*, and *Solenopsis molesta* dominant over *Monomorium minimum*. Each cell of the lattice simulated the probability of each of the up to eight neighboring cells excluding the occupant of the central cell out of the lattice. That is, based on the competition coefficients each resident occupant could be competitively excluded from the simulation to extinction. To reflect the relative size of our model to the original model, we input our values into a 10 by 10 cell lattice to visualize what we theoretically should be able to observe on a scale a fraction of the size of the total 92m by 22m grid. To reflect the edge effects of a natural habitat we set the edges of the model to serve as boundaries and not synchronized with
opposite edges to maintain a flow of individuals from one edge back into the simulation. Stochasticity was also built into the model to replicate different anomalies that are sure to arise in a population over time due to temporal, spatial, or climatic/meteorological changes. Model simulations were run in sequential order of increasing magnitudes from 10 by 10 cell lattice up to 100 by 100 cell lattices in order to contrast scalar patterns of competition.

RESULTS

Field Study

Figure 1. The Sequential Presence of *Solenopsis molesta* in Abby’s Plots

Red triangles represent *S. molesta*, green large circles represent *T. caespitum*, and black bullets represent *M. minima*.

In Abby’s plot we see a slow but gradual increase of colonization on tuna fish baits by *Solenopsis molesta*. In the initial response, *S. molesta* was located seemingly randomly throughout the lattice. As time progressed, more distinct patterns of colonization emerged. This is shown in Abby’s Second Plot where *S. molesta* has become more clumped towards the midline of the plot. By the final response, the ants had clustered and dominated in one general region of the upper-lattice. It is important to note that with every time response, the number of *S. molesta* colonies increased in Abby’s plot.

Figure 2. The Sequential Presence of Ants in Dave’s Plots
Red triangles represent *S. molesta*, green large circles represent *T. caespitum*, and black bullets represent *M. minima*.

In Dave’s plot all three ant species of interest are found at various times. *Solenopsis molesta* is ever present. *Monomorium minimum* is also present at all three times. *Tetramorium caespitum* was only found at the second time response and did not persist to the third response. The presence of *S. molesta* increased throughout the experiment as it colonized more open baits and competed with the other species for dominance over the tuna fish. *S. molesta* appears to be the strongest competitor in this system, as all baits initially discovered by *T. caespitum* in time 2 were dominated by *S. molesta* by time 3. Only one of the baits initially colonized by *M. minimum* was completely taken over by *S. molesta*. *M. minimum* appears to have abandoned bait, assumingly in response to the strong presence of *S. molesta*. There is one point in Dave’s lattice that show a shared bait, which is likely to be in transition from dominance of *M. minimum* to *S. molesta*.

Figure 3. The Sequential Presence of Ants in Iman’s Plots

Red triangles represent *S. molesta*, green large circles represent *T. caespitum*, and black bullets represent *M. minima*.

In Iman’s plot *S. molesta* was the first to colonize the baits. In Dave’s plot *T. caespitum* did not appear to be a strong competitor against *S. molesta*, but Iman’s plots show different interactions between the two species over time. In fact, the number of baits colonized by *S. molesta* decreased over the course of the experiment while the number of baits colonized by *T. caespitum* increased. By the final response the lattice was almost entirely occupied by one of the two ant species, and they appeared to be forming a pattern that split the lattice in half vertically. Competition between species was minimized by the prevention of simultaneously occupying baits. Essentially, strong-holds were created along the sides of the lattice.

Figure 4. The Sequential Presence of Ants in Susan’s Plots
Red triangles represent \textit{S. molesta}, green large circles represent \textit{T. caespitum}, and black bullets represent \textit{M. minima}.

Susan’s plots show a very interesting pattern. \textit{S. molesta} is the first to colonize the plot, but as \textit{T. caespitum} comes in the dominance in the lattice stabilizes. Between time 2 and time 3 roughly the same numbers of baits are occupied. The dominance of \textit{T. caespitum} increased as the dominance of \textit{S. molesta} decreased. Again we saw the two species develop a visual pattern that suggested the ants try to minimize competition by occupying relatively non-overlapping areas of the lattice.

Figure 5. The Sequential Presence of Ants in Bill’s Plots

Bill’s plots were minimally colonized by only \textit{S. molesta} during the initial response. The number of ants continued to increase throughout the duration of the experiment. At time 2 one point was occupied by \textit{T. caespitum}. \textit{T. caespitum} appeared to have abandoned this point by time 3 and came into direct competition with \textit{S. molesta} on an alternative bait.

Figure 6. All Initial and Final Response Plots
Red triangles represent *S. molesta*, green large circles represent *T. caespitum*, and black bullets represent *M. minima*.

Aligning all initial and final response plots shows the general trend of the lattice filling and spatial patterns forming. The graphs show the prevalence of *S. molesta* as a key species in the lattice. *S. molesta* appears to be successful in both the discovery and dominance of the baits. *T. caespitum* is also successful at dominating baits and seems to impede the complete takeover of *S. molesta* in the landscape. When *S. molesta* and *T. caespitum* are both present in the lattice they appear to form fluid distributions which may minimize competition. *M. minimum* was only seen in one plot and was not able to strongly defend its baits against *S. molesta* in that plot.

**Mathematical Model**

The model simulation at the 10 by 10 cell lattice scale consistently resulted in the loss of species within the population and the emergence of one dominant species. The duration of the simulation varied in lengths, but the mean time of simulation was 3 iterations before the emergence of a dominant species and 4 to 5 iterations before the loss of all biodiversity leaving only one species remaining. Neither species was able to dominate the simulation with any consistency, leaving the randomization of founder species as the probable determinant of the dominant species.

The simulations of the next size classes (20 by 20 and 30 by 30 cell lattices) had relatively uniform results with no major contrasts until examination of the 40 by 40 cell lattice. The duration of the 40 by 40 cell lattice simulation was the longest by far, and tended to have far more variation in the emergence of a dominant species. In most cases, the simulations would have entirely different emerging dominant species as well as pattern formations. The correlation with dominant species and founder species in the lattice was less of a determinant factor of outcome and the distribution of species clumps seemed to be a better indicator of how the simulation would conclude. If the spatial conformation of the clumps was in such a way that all neighboring cells of one species clump were occupied by the species possessing dominant competition coefficients, then that subordinate species would be excluded out of the simulation. The edge effects also provided a compounding factor to this mechanism, as the number of neighboring cells that could possibly be inhabited by subordinate was decreased in corners and along the sides of the lattice. This increased the probability that any species with all individuals
clumped in one area would be driven to extinction if all neighboring cells were occupied by a dominant species.

Figure 7. 40 by 40 Cell Lattice Simulations

The 50 by 50 cell lattice was the first size class that was able to sustain biodiversity for any substantial period of time (200 or more iterations). It was also the minimum lattice size class to demonstrate the spatial patterns observed in the 100 by 100 cell simulation that serves as our standard scale for which the model was designed to analyze. The emergence of a dominant species was consistent with the smaller lattice simulations, but the point at which the loss of biodiversity seemed eminent was uniformly observed to be followed by the formation of a spiral conformation around that critical cell cluster. These spiral patterns would then continue throughout the simulation growing in size, followed by fragmentation along edges of the lattice and then reformation as smaller clumps of all three species started smaller spiral patterns throughout the lattice that conformed back into a larger spiral. The stability of this lattice and all larger lattices (which represent a system where all three species can coexist) seemed to be a product of the spiral conformation demonstrated through different stages of the simulation, closely resembling the intransitive loop inherently found in our competition coefficients.

Figure 8. 50 by 50 Cell Lattice Simulations

DISCUSSION
Model simulations of competition that develop intransitive loops suggest that populations can sustain themselves over time. Even when stochasticity is accounted for in these models, declining populations recover. However, this seems to be only possible when considered on a large enough scale, such as in a 100 by 100 cell lattice. On such a plot, declining species are located at edges and appear to be approaching extinction. Yet given a sufficient amount of time, the few remaining individuals at those edges are enough to increase the population. On a half-scale plot, this pattern was also seen. However, when the scale of our study (10 by 10 cell lattice) was used to run the intransitive loop model, one species quickly dominated the entire plot. This domination over the entire plot implies that the other two species will no longer exist within this area.

The inability for declining populations to avoid extinction in small plots has several implications for conservation, especially in regards to habitat loss and habitat fragmentation. Previous research has shown that habitat fragmentation can negatively impact species diversity and abundance (Debinsky and Holt 2000, Steffan-Dewenter et al. 2002, Connor et al. 2000). This can occur because habitat fragmentation results in patch sizes that are too small to sustain a population, an increase in impediments that prevent species from moving between patches in a landscape, and more negative interactions with other organisms at habitat edges (Fahrig 2003). Although some studies show that habitat loss can lead to positive impacts on populations (Tscharntke et al. 2002, Fahrig 2003), our model suggests otherwise. Thus, the consequences of small habitat sizes should be seriously considered during any industry or policy decisions related to the fragmentation of landscapes.

ACKNOWLEDGEMENTS

We would like to thank Senay Yitbarek for his conceptual contributions of the model constructed for the analyses undertaken in this study.

LITERATURE CITED


Levins, R. and Culver, D. 1971. Regional coexistence of species and competition between
rare species. PNAS 68: 1246-1248.


ACARODOMATIA OF WOODY PLANTS OF THE E. S. GEORGE RESERVE

DAVID GONTHIER, CINDY BICK, RACHEL CABLE, LAUREN MALONEY, KATE ZEMENICK, CHRIS DICK

ABSTRACT

Acarodomatia are leaf morphological structures that harbor mites (Acari). There are currently a number of examples demonstrating that mites living within acarodomatia can be beneficial to the plants that host them. Yet there are few detailed investigations describing the distribution of acarodomatia across plant phylogenies. The purpose of this study was to document the species of woody plants within the E. S. George Reserve that have acarodomatia. Forty-six species of woody plants were collected, of which, 14 species were noted to have tufted acarodomatia. Four of these 14 species also were classified to have pocket acarodomatia. A phylogenetic tree of the species sampled showed dispersion of acarodomatia across the phylogeny. However, seven of 14 tuft and three of four pocket acarodomatia fell within the order Fagales. Further analysis revealed acarodomatia were present more often on hairy leaved species than smooth leaved species. Similarly, there were marginally more native species with acarodomatia than exotic species. These results, as well as, other reports within the literature suggest a large majority of woody species have adapted acarodomatia. Further, they suggest the need for more studies on the interactions between mites and plants, as well as, the need for the elucidation of the evolutionary history of these acarodomatia within plants.

INTRODUCTION

Co-evolution between species is one of the hallmarks of the ecology plant-animal interaction (Ehrlich & Raven 1964). For plants that have evolved domatia to house arthropods the relationship may be both strong and mutualistic. For example, domatia baring ant-plants have high mortality or reduced fitness in the absence of their ant partners (Janzen 1966). In these cases aggressive ant partners are important for limiting herbivore and seed predators of their plant partners (Rosumek et al. 2009, Chamberlain & Holland 2009).

Many plants also bare acarodomatia, small tufts, pockets, or pits that house mites (Acari) (O’Dowd & Willson 1991). However, very little is known about function of these domatia and the role of mites on plants. Further, the presence of domatia on plants is still not well documented or described even for temperature latitudes. There is some mounting evidence that mites may play mutualistic roles with plants (O’Dowd & Willson 1991). For instance, on both agricultural and wild grapes (Vitis spp.) fungivorous mites provide indirect resistance to grape infection by a powdery mildew (Norton et al. 2000, English-Loeb et al. 2005, English-Loeb & Norton 2007). The denser the domatia on a given plant leaf, the more mites present and the greater the protection the plant receives from fungal
attack (Norton et al. 2000, English-Loeb et al. 2005, English-Loeb & Norton 2007). Other studies have shown that mites benefit from domatia structures by avoiding predation (Norton et al. 1991), thus for this case it appears mites and acarodomatia baring plants form a defense mutualism. Yet the majority of mite-acarodomatia baring plant interactions remain undocumented. But, before further description of the interactions between mites and plants can be depicted, subsequent documentation of acarodomatia across the kingdom Planta is needed.

Within the temperate zone there are still few in-depth surveys of domatia across woody species. Willson (1991) documented the acarodomatia of 71 Illinois deciduous trees and found 34 (48%) had well-defined domatia and 16 (23%) had “halfway houses” or weakly-defined domatia for a total of 70% of woody plants containing some-sort of acarodomatia. However, there are over 800 species of woody plants in Eastern North America, most of which have still not been described. The purpose of this study was to document the presence of domatia on woody plants of the E.S. George Reserve in Eastern Michigan, USA.

MATERIALS & METHODS

We conducted a survey of the acarodomatia of woody plants within the Edwin S. George Reserve (ESGR) on October 2nd, 2010. The ESGR has been covered by 60% woodlands. Its main vegetation types include grassy old fields, as well as, oak-hickory forests. However, there are many marshy and swampy areas that allow for wetland woody species to also exist within the reserve. To survey the reserve, we opportunistically collected branches from trees along roadways and within forests, grasslands, and swamps. If a branch was out of reach, we used a pole cutter to collect the branch. After collection, we used Bauch & Lomb dissecting microscopes (0.7-3X) to determine if acarodomatia were present on the leaves at vein junctions. If acarodomatia were present, we described whether it should be defined as tuft, pit, pouch, or pocket domatia as described in O’Dowd (1989). The presence of mites within acarodomatia was noted.

After all species had been described for acarodomatia, we created a phylogeny of the species sampled with the program Phylomatic and overlay the presence of the different type of acarodomatia found. Further, a table of all woody species sampled in the current study was created documenting the presence of tuft and pocket acarodomatia.

The data set allowed a limited number of comparisons to be made across functional plant groups. To determine if the presence of hairy trichomes on leaf surfaces was important to the presence of acarodomatia on plants, we compared the number of acarodomatia on plants with hairy leaves relative to plants with smooth leaves using Chi-squared test (SPSS). Additionally, we compared whether the number exotic plants harboring acarodomatia differed from the number of native plants with these structures using a Chi-squared test.

RESULTS

Forty-six species of woody plants were collected, of which, 14 species were noted to have tufted acarodomatia. Four species of the species that had acarodomatia also were classified to have pocket acarodomatia. A phylogenetic tree of the species sampled shows
dispersion of acarodomatia across the phylogeny (Fig. 1). However, seven of 14 tuft and three of four pocket acarodomatia fell within the order Fagales.

Figure 1. Phylogeny of the woody plants sampled within the study. Red circles mark locations where tuft domatia occur. Red boxes mark locations where pocket domatia occur.
There were more species harboring acarodomatia that had hairy leaves than species with smooth leaves ($X^2 = 9.35, P < 0.01$, Table 1). In fact, there were no plants with smooth leaves that had domatia. There were more native species with acarodomatia than exotic species, however because of low sample size this difference was only marginally statistical ($X^2 = 3.02, P = 0.08$, Table 2).

Table 1. The number of smooth and hairy leaved woody plant species harboring acarodomatia.

<table>
<thead>
<tr>
<th></th>
<th>Smooth</th>
<th>Hairy*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acarodomatia</td>
<td>0</td>
<td>14</td>
</tr>
<tr>
<td>No acarodomatia</td>
<td>15</td>
<td>18</td>
</tr>
</tbody>
</table>

*$X^2 = 9.35, P < 0.01$

Table 2. The number of exotic and native species harboring acarodomatia.

<table>
<thead>
<tr>
<th></th>
<th>Exotic</th>
<th>Native</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acarodomatia</td>
<td>0</td>
<td>14</td>
</tr>
<tr>
<td>No acarodomatia</td>
<td>6</td>
<td>26</td>
</tr>
</tbody>
</table>

$X^2 = 3.02, P = 0.08$

Table 3. List of woody species collected and the presence of tuft and pocket acarodomatia.

<table>
<thead>
<tr>
<th>Taxonomy</th>
<th>Acarodomatia</th>
<th>Celastrus orbiculata</th>
</tr>
</thead>
<tbody>
<tr>
<td>Adoxaceae</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sambucus canadensis</td>
<td></td>
<td>Cornaceae</td>
</tr>
<tr>
<td>Viburnum acerifolium</td>
<td></td>
<td>Cornus amomum</td>
</tr>
<tr>
<td>Viburnum trilobum</td>
<td>tuft</td>
<td>Cornus florida</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cornus foemina</td>
</tr>
<tr>
<td>Anacardiaceae</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rhus globera</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Berberidaceae</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Berberis thunbergii</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Betulaceae</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Corylus americana</td>
<td>tuft &amp; pocket</td>
<td></td>
</tr>
<tr>
<td>Ostrya virginiana</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Caprifoliaceae</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lonicera tatarica</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Celastraceae</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
**DISCUSSION**

Our study concluded that acarodomatia were mostly prevalent on woody leaves with hairy surfaces compared to woody leaves with smooth surfaces (Table 1). Several studies (English-Loeb 2007, English-Loeb et al. 2005, Norton et al. 2001, Rozario 1995, O’Dowd & Willson 1991, Willson 1991) have documented the presence of fungivorous and predatory mites on woody plants and have also suggested mutualistic associations between these mites and their plant hosts. Mites use various forms of plant structures such as invaginations, vein junctions, hair tufts, and pockets to construct acarodomatia. These structures are too small for most insects to enter (Ruberson 1999).
Walter (1996) suggested that leaf architecture and topography is integral to the presence, predator-prey interactions, and the abundance of mites on a leaf. It was also suggested that mite abundance varied according to the degree of pubescence on the foliage (Walter 1996). Tomentose leaves provide refugia from the sun, rain, and predators (Ruberson 1999). Additionally, acarodomatia also have relatively high humidity that prevents desiccation of mite eggs (Ruberson 1999). Therefore, it should be expected that foliages with hairy surfaces should have more acarodomatia compared to foliages with smooth surfaces.

However, not all plants with hairy surfaces had acarodomatia. There were more hairy plants without acarodomatia than hairy plants with acarodomatia (Table 1). Willson (1991) reported that more than 70% of the tree species sampled from deciduous forests in central Illinois had acarodomatia and more than 50% of those tree species were occupied or had evidences of occupation by mites. For each species (29 total), Willson (1991) collected 5 leaves on up to 6 individuals per site. In contrast to our study, leaves were collected from 46 tree species from only 1 individual. Walter (1996) has suggested that mite numbers and acarodomatia vary widely between adjacent trees depending on the level of hairiness. A lot of the trees present at E. S. George Reserve grew in stands. Therefore, sampling from one individual would not provide a representative sample for elucidating how common these foliar structures are and what plants produce them.

Nonetheless, from a phylogenetic standpoint, half of the species with acarodomatia were within the order Fagales (Figure 1). Only a few studies (Fiala & Maschwitz 1992) focus on the adaptive evolution of acarodomatia, but not necessarily on the phylogenetic relationship of acarodomatia with a specific plant taxa. It may be that acarodomatia is not specific to any specific plant order or family, as some studies have reported that trichomes and thorns on leaf surfaces are induced by herbivory (Dalin et al. 2008, Agrawal et al. 2002, Traw & Dawson 2002, Ruberson 1999). However, if acarodomatia is indeed a plant-mite mutualistic phenomena then it would be of interest to investigate the phylogenetic relationship between plants and acarodomatia.

Interestingly, we also found no acarodomatia on any of the exotic plant species that were collected. Although the significance of the relationship between the presence of acarodomatia and exotic plant species was marginal, the fact that acarodomatia was absent from exotics should not be ignored. The absence of acarodematia from exotic species may mean that it is subjected to very little or no herbivory and therefore, mites have not had sufficient time to adapt to these plant species. Research by McMurtry and Croft (1997) has suggested that in some instances indigenous mites do exploit exotic leaf structures based on leaf structure specialization rather than plant fidelity. *Schinus terebinthifolius*, also known as the Brazilian peppertree is an advent species to Florida and was originally documented as having little or no mite associations. Now, a wide array of native mites could be found associated with it (Wiggers et al 2005).

Goeden (1974) and Strong et al. (1984) also suggested that communities with exotic plant species are usually characterized by a lack of native faunal assemblages specifically associated with a plant in its non-native range. Furthermore, many studies suggest that greater diversity and abundance of natural herbivores would occur in a plant’s natural environment compared to its introduced range. Therefore, if presence of hairy phylloplanes is indeed induced by herbivory as previous studies (Dalin et al. 2008,
Agrawal et al. 2002, Traw & Dawson 2002, Ruberson 1999) have suggested, then this is a probable explanation for the absence of acarodomatia from exotic plants foliage.

There were a number of limitations to the study conducted. First, for most woody species surveyed only one individual was examined. In some species of plant, like grapes (*Vitis* spp.) there is large variation in the density of domatia across cultivars (English-Loeb et al. 2005) and this variation is likely observed across genotypes in most plants. Another limitation is clearly the difficulty in defining what is a tuft domatia. Given that tuft domatia are made up of pubescent hairs, usually on leaves that are generally hairy, it is difficult to distinguish some domatia from leaf hairs themselves. This fact is clear when the lists of domatia baring plants documented in this study is compared to others. Robyn Burnham’s (Unpublished) survey marked American elm as a tuft and pocket domatia species, whereas here we did not define it as a domatia species. Additionally, Willson (1991) did not define black cherry (*Prunus serotina*) as a domatia species; however, in the current study and in R. Burnham (Unpublished) black cherry was recorded as a tuft domatia species. These facts strongly encourage the development of a more rigorous method of defining domatia, especially those of the tuft variety.

In conclusion, here we document the distribution of acarodomatia across 46 woody plants species within the E. S. George reserve. The 14 species with acarodomatia provide further evidence that these structures are a ubiquitous feature to plants across the temperature region, but suggest the need for further investigations to document the ecological and evolutionary role of mites on domatia baring plants.

ACKNOWLEDGEMENTS

Thanks to Ivette Perfecto, John Vandermeer, and David Allen for help with project design and logistics.

LITERATURE CITED


triphasic interactions among grapevines, a fungal pathogen, and a mycophagous


Evolution 18: 586-608.

Fiala, B. and U. Maschwitz. 1992. Domatia as most important adaptations in the
evolution of myrmecophytes in the paleotropical tree genus *Macaranga*
(*Euphorbiaceae*). Plant Systematics and Evolution 180: 53-64.

Goeden, R. D. 1974. Comparative survey of the phytophagous insect faunas of Italian
thistle, *Carduus pycnocephalus*, in southern California and southern Europe

Janzen, D. H. 1966. Coevolution of mutualism between ants and acacias in Central


invertebrate fauna on broom, *Cytisus scoparius*, in two native and two exotic

enemies: leaf domatia protect beneficial mites from insect predators Oecologia
126: 535-542.

and foliar pathogens: leaf domatia mediate tritrophic interactions in grapes.

O’Dowd, D.J. 1989. Leaf domatia and mites on Australasian plants: ecological and

TREE 6: 179-182.

Pemberton, R.W. and Turner, C.E. Occurrence of predatory and fungivorous mites in leaf


SPATIAL DISTRIBUTION OF BARBERRY IN E. S. GEORGE RESERVE

JOHN GUITTAR, SUSAN CHENG, SAHAR HAGHIGHAT, BINBIN LI, ABIGAIL HYDUKE, WILLIAM WEBB, JOHN VANDERMEER

ABSTRACT

Highly invasive plants can drastically alter the landscape of a forest community on a minimal temporal scale. Japanese Barberry (Berberis thunbergii) was introduced to the E. S. George reserve ca. 1970 and has proliferated rapidly since then, making it a good model to study recent invasions. One important aspect of plant invasions is their spatial organization; certain distributions may indicate certain characteristics of its spread, or the stage of invasion. A power-law distribution can typify a self-organizing system, for example. In this study we correlate Japanese Barberry spatial distribution at a 10m x 10m resolution. In agreement with previous work we find Japanese Barberry to follow a power-law distribution of cluster size vs. frequency, although this correlation may be misleading.

INTRODUCTION

Invasive plants can drastically alter the landscape of a forest community on a minimal temporal scale. The life history of a species may inherently leave it predisposed to being able to out-compete native flora for resources and establish itself (Herron 2007, Hamilton 2005). The prevalence of plant invasions can be tied in many cases to species that are introduced primarily for aesthetic reasons or to serve ornamental/decorative purposes before the properties of that species can be assessed for possible ecological complications. Japanese barberry (Berberis thunbergii) has been identified as a highly competitive introduced species because of its ability to utilize resource poor environments (Harrington 2004). Barberry can survive in as low as 4% of direct sunlight (Silander and Kleipis 1999). Lastly, Barberry produces a large number of bird dispersed berries (Kendal 2004), but can also reproduce clonally. This suite of characteristics makes Barberry a particularly strong invader. The Berberis invasion of the E. S. George Reserve in Livingston County, MI is a good candidate to understanding invasion in a forest understory system.

An invading species colonizing a habitat creates patterns in time and space. Identifying and understanding these spatial and temporal patterns can give us clues to the invader’s past and present distributions, as well as the underlying mechanisms driving its spread. For example, one commonly observed pattern in the size structure of populations is a power-law distribution, a signature of self-organization (Kefi 2007). In this case, if Japanese Barberry demonstrated a linear log-log relationship between size and frequency of patches, this would be a power-law distribution, and would suggest the population is self organizing. That is, the distribution of Barberry patches size and number would be consistent regardless of scale. Self-organization can be representative of a system that is density-dependent, as opposed to environment or niche dependent.

Previous studies (Field Ecology 2009, 2010) have found Japanese Barberry spatial distribution to follow a power-law distribution at larger scales (~16 ha). In this study we
investigate the spatial organization and potential invasion patterns of Japanese Barberry at a higher resolution (2 ha). Further evidence of a power-law distribution would strengthen the hypotheses that barberry have established a scale-free distribution. In addition, we explore the possibility that Japanese Barberry distributions are not self-organizing by comparing its density to the heterogeneous forest and canopy composition above. Specifically, we look for correlations between Barberry and Black Cherry, Witch Hazel, and Red Maple tree distributions. If linkage with tree distribution is strong, we would suspect that an environmental variable is governing Barberry distribution more strongly than self-organizing processes.

METHODS AND MATERIALS

Two hectares were selected using data from previous surveys on the distribution of Berberis thumbergii in the Big Woods of the E.S. George Reserve. Areas were selected on the basis of higher B. thumbergii density and diverse canopy tree species distribution. Hectares were subdivided into ten 100m transects and visually estimated for percent B. thumbergii coverage (0-100% of ground cover) by a pair of individuals in 10m increments. Both individuals determined the percentage individually and then calculated the mean value. The two teams of estimators were constantly rotated to avoid estimation divergence. Data was organized using Excel then compared to Big Woods tree species spatial data (Allen 2008) to investigate spatial relationships. Data from previous Field Ecology surveys in 2009 and 2010 were used to compare self-distribution of barberry.

RESULTS

The distribution of B. thunbergii in the two plots surveyed in the Big Woods can be seen in Figure 1. Few large patches of B. thunbergii were found. Instead, the sampled area was primarily characterized by many small patches of B. thunbergii. Accordingly, the distribution of B. thunbergii patch size appeared to follow a power-law function (Figures 2 and 3). When overlain over a topographic map, B. thunbergii appeared to follow no pattern with topography. However, a trend was seen between barberry patch size and the density of trees with a girth at breast height of 10 centimeters or greater. B. thunbergii patches tended to be smaller in areas that included a higher number of witch hazel (Hamamelis virginiana), red maple (Acer rubrum), and black cherry (Prunus serotina) (p<0.0001). Deeper analysis of tree density and B. thunbergii data showed that this negative correlation remained with B. thunbergii patches and A. rubrum (p<0.01) and H. virginiana (p<0.001) (Figures 5-8). However, this correlation did not hold true for B. thunbergii and P. serotina (p>0.05) (Figures 9-10).
Figure 1: Distribution of *B. thunbergii* in two plots in the Big Woods. Bubble size represents the percentage of barberry cover in each 10 x 10 m subplot.

Figure 2: Histogram of *B. thunbergii* patch size

Figure 3: The frequency of *B. thunbergii* patch sizes on a log-log scale
Figure 4: *B. thunbergii* patch size versus total number of trees

\[ y = -1.3366x + 28.44 \]
\[ R^2 = 0.1174 \]

Figure 5: Distribution of the number of *A. rubrum* in sampled plots.

Figure 6: *B. thunbergii* patch size versus number of *A. rubrum*.

\[ y = -1.0621x + 19.014 \]
\[ R^2 = 0.0527 \]
\[ p < 0.01 \]
DISCUSSION

In agreement with studies on Japanese Barberry distribution (Field Ecology 2009, 2010) we find Barberry to significantly follow a log-log distribution at a 10m x 10m resolution (Figure 2; Figure 3). This lends credence to the hypothesis that Japanese Barberry is operating as a self-organized system. Yet this may not be the case. With limited data, other trends can superficially resemble a log-log relationship which could confound this conclusion. For example, expanding populations can display a lognormal relationship, caused by the normal distribution of their growth rates. A characteristic difference in a lognormal graph is tapered curve into the frequency axis. This could be envisioned in Figure 3. Secondly, the Japanese Barberry invasion could still be in flux and operating according to different growth and dispersion dynamics. Further data collection and distribution modeling is necessary to explore alternative possibilities.

The rapid invasion of Japanese Barberry since its arrival to the E. S. George Reserve ca. 1970 may be constrained, or at least negatively correlated, with canopy coverage (Figure 4). While co-occurrence with Black Cherry (*Prunus serotina*) was random, negative correlations with red-maple (*Acer rubrum*) and witch-hazel (*Hamamelis virginiana*) presence were apparent. Since Witch Hazel can form dense mid-canopy clumps (Figure 7) which may stifle Japanese
Barberry through light limitation. It would be interesting to see if other shrub-like species persisted in witch hazel groves in the reserve.

AKNOWLEDGMENTS

We would like to thank Dave Allen for assisting in data collection and analysis.

LITERATURE CITED


Kendal, W. Taylor’s ecological power law as a function of scale invariant exponential dispersion models. Ecological Complexity 1:193-209


LIANA DIVERSITY AND ABUNDANCE FOLLOWING SUCCESSION IN THE
E.S. GEORGE RESERVE

ABIGAIL HYDUKE, ROBYN BURNHAM, SUSAN CHENG, JOHN GUITTAR,
SAHAR HAGHIGHAT, BINBIN LI AND WILLIAM WEBB

ABSTRACT

Lianas are integral members of both temperate and tropical plant communities. They can influence tree reproduction and are themselves susceptible to presence or absence of trees. Research on lianas from tropical and subtropical forests suggests that they are quick colonizers of openings, which result following a disturbance. However, they are also well-established members of mature forests. We surveyed and censused lianas in a secondary forest and an abandoned agricultural field. We compared our data with records from a nearby primary forest to determine if liana abundance and diversity shifted predictably with succession. Our results, although inconclusive due to small sample size, imply that secondary forests will have greater diversity than either older forest or regenerating agricultural fields. Our results also reveal a shift in species composition and in climbing type, with the highest diversity of climbing type found in the primary forest.

INTRODUCTION

Lianas are important components in plant communities. Most are shade tolerant species that wait for favorable light conditions to respond with high growth rate (Leicht and Silander 2006). Much research has been carried out in the tropics, where richness and abundance of lianas is highest. However, lianas in temperate forests also play an influential role in shaping plant communities and influence forest regeneration (Ladwig and Meiners, 2010). Climbing mechanism is one of the factors suggested to influence liana impacts on regeneration because it is an indicator of the type of host trees that lianas climb (Penalosa, 1982). Twining lianas require small trees, which they can curl around, while those with tendrils can climb any trees large enough to support their weight including late-successional species (Carter and Teramura, 1988). Beyond choosing host according to size, diameter, and bark texture, lianas can influence regeneration. By causing trunk constriction and removal of bark, shoots, and buds, and forming shade canopies they, in effect, prune the host tree and influence resource allocation (Laura and Scott, 2010). Recent research has shown that lianas have affinity for trees colonizing disturbed
areas and have higher abundance following disturbance (Allen et al. 2007). Lianas could persist in mature forests but are found abundantly along a forest edges (Ladwig and Meiners, 2009).

Previous studies in tropical and subtropical forests have shown that disturbances such as habitat fragmentation, gap formations, and logging can result in higher liana abundance and diversity (Laurance et al. 2001, Ding and Zang 2009, Malizia et al. 2010). The goal of our project was to determine how liana diversity in temperate ecosystems would change as time passed after a disturbance. We predicted that liana diversity would be highest in secondary forests because a semi-open canopy and the availability of woody hosts for climbing would provide opportunities for many lianas to establish. In contrast, the lowest diversity would be found in primary forests because the few species that established themselves earlier in the forest history would dominate (Ladwig and Meiners 2010). We also predicted that diversity would be low in recently abandoned fields due to the lack of woody material for lianas to climb and thus, the resulting competition that would exist amongst a few opportunistic species.

**METHODS**

Our group sampled two sites on the E. S. George Reserve in Livingston County, MI that underwent disturbance at different times. The first site was located in a secondary forest at least 25 years old (according to a tree core we took) near a portion of the West Woods that experienced a tornado in 1990 (Vandermeer, personal communication 2010). The second site was an open field located in the AirField (42°27'29.4” N, 84°00'45.9”W)

At each site, we censused five contiguous 10 m x 10 m plots. For each plot, seven investigators surveyed and censused all species of lianas. We identified each liana species and estimated its percentage ground cover. We also recorded the number of trees found in the plots, their species, and their diameter breast height (DBH) if ≥ 15 centimeters. For each tree, we identified the species and number of liana individuals growing on each tree. In addition, we measured liana stems if they were thicker than 0.5 cm. We later compared our data with a similar study from a primary forest of the Big Woods plot of the E.S. George Reserve to compare species composition and abundance (Burnham, *unpublished*).

**RESULTS**

Percent cover of climber species is shown in Figure 1. Secondary Forest and Old Field had similar species richness and were dominated by a single but different *Rubus* species, both with a “scrambling” climbing strategy. While percent cover was disproportionately dominated by one species, presence/absence data indicates a consistent distribution of species (Figure 2; Table 1). The Big Wood was not included in the first comparison because of differences in sampling techniques. Burnham recorded frequency in 50 10 x 10 m cells, rather than percent cover in each cell.
Figure 1: Percent cover of climbers by species, in order of decreasing frequency.

![Secondary Forest](image1)
![Old Field](image2)

N=5

Figure 2: Presence/Absence of climbers by species, in order of frequency across the 5 plots.

![Secondary Forest](image3)
![Old Field](image4)

N=5

![Big Woods](image5)

N=50
Figure 2 demonstrates a similar distribution in all three sites for species presence or absence data. The differences are more apparent in the species composition. *Parthenocissus quinquefolia* is found consistently in all three sites. Different species of *Rubus* are also highly coincident between the sites.

Species richness curves for Secondary Forest and Old Field did not reach an asymptote during our sampling, suggesting we would likely encounter more species with continued sampling. Using EstimateS MMMean calculations, predicted richness for Secondary Forest is 10.66 and Old Field is 11.75, but we have low-confidence in these estimates because of the small sample sizes. Richness curves are compared with those from Big Woods data from 2007 (Burnham, *unpublished*) in Figure 3.

Figure 3: Species richness curves for climbers in three habitats. Solid and dashed lines (+/- 95% confidence for Big Woods) were calculated in EstimateS. Confidence curves for Secondary Forest and Old Field have large overlap and are not shown.
Species are arranged by family in Table 1, Smilacaceae and Dioscoraceae are listed first as Monocots. Thereafter, the list is arranged by families that both contained more species, and were more commonly encountered. Monocot vines were only found in the Big Woods site. Rosaceae is the largest family found across all three sampling sites, with six species.
Species that exhibit more than one climbing strategy were recorded for each strategy in Figure 4. Adventitious roots and adhesive pads are consistent across all three sites. Tendril climbers and apical twiners increase in primary forest as scramblers decrease. An over-all increase in diversity of climbing types is seen in the primary forest.

**DISCUSSION**

The data agreed with our hypothesis that the rapid response of vine growth in secondary forest would lead to increased biodiversity and higher species richness, compared to undisturbed, older forests (Ladwig and Meiners 2010). Our data can serve as a suggestion of several trends of liana growth in regenerating and old-growth deciduous forests. We found one more species of vine in the secondary forest (West Woods Plot) than in our recently disturbed site (Old Field). The variety of climber types within the secondary forest community does reflect the microhabitat, differences as there is more diversity found within the primary forest plots. The climbing mechanisms between old-field and secondary forest are more similar and may be attributed to the prevalence of scrambling *Rubus* spp.

Open canopy structure allows for differential success of different strategies because success is not only dependent upon the taller trees towards the canopy (Leicht and Silander 2006). However, the near decade that has passed since the tornado disturbance altered the landscape and canopy structure of our plot may not have been enough time for seed dispersal of other vines outside of the local population to reach the area now suitable to growth of most species. Vines that are not effective climbers are usually out-competed once the canopy layer
fills in throughout the forest. The lowered percent coverage will now expose many of the
ground-dwelling individuals to direct sunlight necessary to facilitate growth. One of the
underlying factors affecting why this mechanism cannot take place is that most of the canopy
layer in this forest is filled with the exception of the segment in the direct path of the tornado.

The biomass of vines in older forests would be expected to be relatively larger than vines
in disturbed forests, reflecting the longer growth period for an individual. The population may
also have a lower capacity of species richness due to the denser layers of canopy cover. While
our species richness curves (Fig. 3) suggest this would be the case as 9 species were found in 5
cells of the secondary forest while only 12 were found in 50 cells of primary forest, a comparable
sample size is necessary to confirm this projection. I am certain that you should subsample my
50 cells and see what you get. There are several programs to do this subsampling – why don’t
you do that for species diversity?

Complications in determining why we had such similar biodiversity may be found within
the theory itself. Recent studies conducted in temperate forests along canopy coverage gradients
(Gianoli et al. 2010) imply that the noted pioneer qualities of vines may be a function of other
factors. Regardless, the observed dominance by the Rubus congener was approximately the
same in both sites. The percentage of liana coverage in both plots was similar, leading us once
again to re-examine the hypothesized pioneer-system of vines. This sounds interesting as an
idea and probably should be discussed more, but I don’t know if that will be worth it without a
lot of data from the literature.

ACKNOWLEDGEMENTS

Special thanks to Dave Allen, Ivette Perfecto, John Vandemeer and Field Ecology 2010.
LITERATURE CITED


AN INVESTIGATION OF FLIGHT INITIATION RESPONSE IN VARIOUS ANIMALS AT THE E.S GEORGE RESERVE

MINGQI LIU, ANA CHARA, RAFAEL D’ANDREA, KASSANDRA SEMRAU, IMAN SYLVAIN AND JOHANNES FOUFOPoulos

ABSTRACT

It is important for the prey animal to decide at what distance to flee when a potential predator is nearby. The optimal flight initiation distance (FID) is determined by complicated factors such as the detecting distance (DD), the distance to the refugia (DR) and both the predators and the prey animals fitness and ability to move. Environmental factors such as the easiness of approaching may also influence FID. In this study, we try to explore the factors that may contribute to FID by examining animals in E.S. George Reserve. Of the 22 observations, 8 species were spotted. There is significant correlation between FID and DD and between FID and DR. However, there are no significance differences of FID for either different obstacle levels or for volant and terrestrial species. There is perhaps artificial effect for the perfect relationship between DD and FID, as DD maybe actually is not the distance for the animal to detect us, but it is when we thought it spot us. The best model is that DD and DR both explain FID, suggesting that the prey may take more factors into consideration when decide to flee. FID is important for wildlife management as it helps to set a baseline for determining buffer zones.

INTRODUCTION

The process of natural selection has led to developments in animal behavior that are necessary for survival. One such key behavior is fleeing when feeling threatened or anxious, especially in predator-prey dynamics. A widespread anti-predator response in animals involves fleeing from approaching predators until attaining a safe distance or entering a hidden refuge (Lagos, 2009). The behavior of flight from predators involves both costs and benefits. The benefit is escaping capture, and the cost is a reduction in fitness. Time spent fleeing reduces the amount of time an animal can spend foraging or engaging in other beneficial activities (Lagos 2009). Because the time required reaching refugia increases with the distance to refuge, animals must make decisions as to when to flee, and at which distance to initiate their flight. Thus there is an optimal flight initiation distance (FID). This is the first point in a predator’s approach when the benefit of flight exceeds the cost (Dill 1989).

Previous studies on fish, lizards, birds and small mammals show that optimal flight initiation distance is explained by many factors. Some of these factors include predation risk assessment, direction of predator approach, predator starting distance, the speed of the predator, and the predator’s body size (Lagos 2009). Other factors that influence optimal FID take into account group dynamics, such as the number of predators (Lagos 2009) and the number of prey present. In fact, social animals or clustered animals are thought to display ‘early detection’ and ‘risk dilution’ which allow them to delay FID (Martin 2005). Although clustered animals may be buffered
against predator, solitary species may run greater risks when determining optimal FID.

The goal of this experiment was determine whether there was any correlation between a prey animal’s detection distance, flight initiation distance, its distance to refugium, and the type of refuge sought. We attempted to investigate these relationships between different species, specifically comparing volant (flying) and terrestrial animals. We hypothesized that volant species would be more tolerant of our approach, as its escape would be easily facilitated by flight to a near-by tree. Following a similar logic, we predicted that terrestrial species would flee quickly and would escape to refugia from a greater flight initiation distance. We anticipated that the distance between the detection distance and flight initiation distance in terrestrial animals would be smaller than in flying animals. We also predicted that terrestrial animals would seek refuge in burrows or under logs, and flying animals would seek refuge in other trees.

METHOD

The study took place in three different forested areas of the George Reserve. Groups of two people walked in semi linear transects along the different sites looking for animals, once an animal was detected one person walked slowly and silently towards it until it ran away. We recorded: Detection Distance (DD: distance between the animal and the observer when the animal was detected), Flight Initiation Distance (FID: distance between the animal and the observer when the animal ran away), Distance to Refuge (DR: distance between the spot where the animal was observed in first place and its refuge), obstacle clearance (qualitative assessment of the level of difficulty to approach the animal, into three classes), the type of escape (distinction between flying and terrestrial animals) and the species name of all the individuals detected.

We performed regression analysis in order to test the effect of DD and DR in the FID. Also we compared the FID of individuals with different escape types and different levels of obstacle clearance. Statistical analyses were carried out using Microsoft Office Excel 2007 and R version 2.11.1.

RESULTS

Over a 3-hour period, 22 observations were made, totaling 5 species of birds and 3 species of mammals. Table 1 displays the Fleeing Initiation Distance (FID) for each species (averaged in case of more than one sighting).

Table 1. Species-averaged FID . *Based on 9 observations. **Based on 3 observations

<table>
<thead>
<tr>
<th>Species</th>
<th>FID</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chickadee**</td>
<td>2.9</td>
</tr>
<tr>
<td>Morning dove</td>
<td>6.0</td>
</tr>
</tbody>
</table>
Obstacle clearance could not be shown to correlate with FID in either a clear or significant way, either between or within species (Tables 2 and 3). Thus, our hypothesis that animals would tolerate closer distances in more obstructed paths is not supported. We also did not find support to the idea that volant animals would allow for closer proximity than terrestrial animals (Table 4). However, were our samples larger, our standard errors could have been small enough that the difference between the means seen in Table 4 could prove significant (assuming, of course, that these means represent the true means).

Table 2. Mean FID for different obstacle clearance levels.

<table>
<thead>
<tr>
<th>Obstacle level</th>
<th>Mean</th>
<th>Std error</th>
</tr>
</thead>
<tbody>
<tr>
<td>1*</td>
<td>5.2</td>
<td>3.3</td>
</tr>
<tr>
<td>2**</td>
<td>8.9</td>
<td>5.2</td>
</tr>
<tr>
<td>3**</td>
<td>14.5</td>
<td>9.3</td>
</tr>
</tbody>
</table>

*Based on 6 observations. **Based on 8 observations.

Table 3. Averaged FID for different clear levels in chipmunks.

<table>
<thead>
<tr>
<th>Obstacle level</th>
<th>Mean</th>
<th>Std error</th>
</tr>
</thead>
<tbody>
<tr>
<td>1*</td>
<td>5.1</td>
<td>3.2</td>
</tr>
<tr>
<td>2**</td>
<td>12.7</td>
<td>3.8</td>
</tr>
<tr>
<td>3*</td>
<td>6.6</td>
<td>2.6</td>
</tr>
</tbody>
</table>

*Based on 3 observations. **Based on 4 observations.

Table 4. Averaged FID for different escaping modes.

<table>
<thead>
<tr>
<th>Type</th>
<th>Mean</th>
<th>Std error</th>
</tr>
</thead>
<tbody>
<tr>
<td>Terrestrial*</td>
<td>10.9</td>
<td>7.3</td>
</tr>
<tr>
<td>Flying**</td>
<td>8.2</td>
<td>7.7</td>
</tr>
</tbody>
</table>

*Based on 14 observations. **Based on 8 observations.

Figure 1. Plot and Linear regression of our data on FID versus DD.
A multivariate linear analysis can be done on the subset of observations containing data on distance to refugia (14 of the 22 sightings). A comparison of models can thus be made in order to determine whether the set “Distance to refugia” + “Detection distance” is better than either of these predictors alone. The results are given on table 4. It can be seen that the model with both predictors works best among all three for that subset of our data, as its AIC index is the smallest and it explains a bigger part of the variance than the other two. Both predictors correlate positively.
with FID.

| Model            | AIC | $R^2$ | DD | $Pr(>|t|)$ | Ref | $Pr(>|t|)$ |
|------------------|-----|-------|----|------------|-----|------------|
| FID $\sim$ Ref   | 78.1| 0.49  | X  | X          | 0.99| 0.005      |
| FID $\sim$ DD    | 66.6| 0.77  | 0.77| 0.00003    | X   | X          |
| FID $\sim$ DD+Ref| 61.3| 0.87  | 0.62| 0.0002     | 0.49| 0.02       |

**DISCUSSION**

The presence of obstacles between predator and prey did not correlate with FID (flight initiation distance), suggesting that prey do not consider how the predator will approach when deciding when to flee. It may be that the prey we sampled weren’t able to distinguish between what would be an obstacle for a large predator and an obstacle for themselves. Despite the correlation between detection distance and FID, we don’t put much faith in our assessment of it. Detection distance in our study was somewhat of a misnomer. It’s highly unlikely that we, the predators, observed the prey before they were aware of our presence. By the time we were aware of the location of the prey, we were already well within the detection distance at which the prey first observed us. Distance to refugia was also correlated with FID. If prey has to travel only a short distance before it feels safe again, it would likely tolerate the predator in a closer range. The further away the refugia, the less distance the prey would tolerate between itself and the predator before fleeing. Distance to refugia and detection distance together were excellent predictors of FID, more so than each alone. This suggests that prey take into consideration multiple factors when deciding when a predator gets too close for comfort.

To maximize their fitness, individuals need to optimize foraging and anti-predator behavior (Ydenberg & Dill, 1986). Being able to make successful decisions about anti-predator behavior is critical, and requires the interpretation of an individual’s surroundings. Anti-predator behavior depends to an extent on the costs and benefits of alternative escape decisions (Bonenfant & Kramer, 1996). Individuals must make escape decisions based on a variety of factors. For example, anti-predator behavior in birds was found to be dependent on intruder starting distance (Blumstein, 2003). While our data agrees with the findings from Blumstein’s paper, our study was not conclusive due to discrepancy between the time the prey detected predator presence and the time the prey visibly moved or called out in warning, alerting the predator to the prey.

Information pertaining to flight initiation distances is highly relevant to wildlife management and conservation because it establishes a baseline distance for buffer zones to prevent the disturbance of wildlife by humans (Rodgers & Smith 1995, Blumstein, 2003). Future studies should focus on flight initiation distances as they apply to individual species, particularly those that are of interest in conservation.

**ACKNOWLEDGEMENTS**
We would like to thank the coordinators (David Allen, Ivette Perfecto and John Vandermeer) of this course for their encouraging feedback.

LITERATURE CITED


HEALTH AND BIODIVERSITY OF THE HONEY CREEK WATERSHED
KASSANDRA SEMRAU, ANA CHARA, RAFAEL D’ANDREA, IMAN SYLVAIN, MINGQI LIU, CATHERINE RISENG

Abstract: In this study, we examine both biotic and abiotic factors in order to establish stream profiles for each of three sites located within the Honey Creek Watershed. We characterized the habitats of each stream in order to determine their physical and chemical states. We collected benthic macroinvertebrates to determine biodiversity measures and compared all characters to determine the overall health of the streams.

INTRODUCTION

Streams and rivers have an extremely important role in the biosphere as conduits for water, nutrients, sediments, woody debris, and biota from the continents to the sea (Pompeu et al. 2005). These ecosystems also provide numerous benefits to mankind like water supply for domestic, industrial and agricultural uses, harvestable organisms, hydropower, waste disposal, navigation, recreational enjoyment, and spiritual fulfillment (Allan and Castillo 2007). Rivers also support unique and complex ecological communities and often influence the structure and functioning of the surrounding terrestrial ecosystem (Pompeu et al. 2005).

Each freshwater body has an individual combination of physical and chemical characteristics that are determined by the climatic, geomorphological and geochemical conditions prevailing in the drainage basin and the underlying aquifer. Characteristics such as total dissolved solids, conductivity and redox potential, provide a general classification of water bodies of a similar nature. Mineral content, determined by the total dissolved solids present, is an essential feature of the quality of any water body resulting from the balance between dissolution and precipitation. Oxygen content is another vital feature of any water body because it greatly influences the solubility of metals and is essential for all forms of biological life (Chapman 1996).

The condition of the river ecosystem is also largely determined by the impact of the human activities in its watershed. Flow variability and sediment delivery to streams are strongly influenced by the impervious surfaces and storm drains of urban areas, the channelized streams and field tiles of agricultural areas, and logging practices and road building in areas of forest harvest. Habitat degradation is likely to be manifested in changes to flow, benthic habitat conditions, and riffle–pool integrity (Allan and Castillo 2007).

Stream assessment is the overall process of evaluation of the physical, chemical and biological nature of water in relation to natural quality, human effects and intended uses, particularly uses which may affect human health and the health of the aquatic system itself (Chapman 1996). The utility of invertebrates in stream assessment has been largely recognized, and is based in the fact that different invertebrate taxa tolerate organic pollution to a lesser or greater extent and that their differing responses can be used to indicate water quality (Hodkison and Jackson 2005).
In this study we assessed physical characteristics, water chemistry and biological condition of three small streams located within the Honey Creek Watershed near the E.S. George Reserve in order to compare the overall status of the sites and relate it to their land use patterns.

MATERIALS AND METHODS

Data were collected from our study sites: No Name Creek (Site one), Putnam Creek (Site 2) and Country Drain No. 7 (Site 3). We measured physical characteristics including water temperature, conductivity (µS/cm) and dissolved oxygen (mg/l). We measured the width of the stream and chose five points along its width to measure depth and flow velocity. We then evaluated the site with respect to structural characteristics including basin land cover, riparian width, riparian structure/composition, bank stability, shading, channel alternation, embeddedness, benthic silt cover and water appearance. We scored these characters by assigning a value on a scale of 0 to 20 (5 for poor, 10 for marginal, 15 for suboptimal, and 20 for optimal). We collected benthic invertebrates from each site using dip nets to sample from various locations in each stream and then identified them to family.

RESULTS

The three streams differed greatly in width and flow, and all three measured very low flow relative to their yearly flow regime (Fig 1, Table 1).

Figure 1. Stream depth and velocity profiles
Table 1. Flow of the three streams, in liters per second. The flow was calculated by $\frac{v(x)D(x)}{W}$, where $v(x)$ is the fitted velocity curve, $D(x)$ is the fitted depth curve, and $W$ is the maximum distance from the left margin (Fig 1). The third column contains the flow for the 5th and the 90th percentile (Riseng, pers. comm.) (the $p^{th}$-percentile indicates the value below which the stream flow is measured at $p$% of the time).

<table>
<thead>
<tr>
<th>Site</th>
<th>Flow (L/s)</th>
<th>5th-percentile</th>
<th>90th-percentile</th>
</tr>
</thead>
<tbody>
<tr>
<td>No Name Creek</td>
<td>8.1</td>
<td>19.0</td>
<td>171.8</td>
</tr>
<tr>
<td>Putnam Creek</td>
<td>61.9</td>
<td>48.7</td>
<td>238.1</td>
</tr>
<tr>
<td>County Drain No. 7</td>
<td>1.6</td>
<td>24.0</td>
<td>230.2</td>
</tr>
</tbody>
</table>

The physicochemical measurements are summarized in the appendix. The average score is shown on Table 2. No Name Creek scored low, bordering on marginal status, while the other two sites are closer to optimal conditions. The table also shows urban and agricultural land use of the watershed. While County Drain No. 7’s watershed is almost twice as managed for agriculture, urban areas are four times more prevalent on No Name Creek’s watershed. The fact that the latter seems to be in much worse shape than the other two might be an indication that urban spread is more important a cause of disturbance in streams than agricultural use.

No Name Creek is seen to consistently score lowest in most of the indices we calculated. It clearly has the poorest results for EPT, surface dependent organisms, and order diversity. Once again, Putnam Creek and County Drain No. 7 show qualitatively similar results, with the former showing better performance on EPT and family richness and the latter faring better on tolerance and order diversity.

Table 2. Various indices calculated from data on insect abundances taken from the streams. Y/N: present/absent. Urban and Agro show percent watershed land use. EPT is the total number of families from orders Ephemeroptera, Plecoptera and Trichoptera. Surf Dep is the total number of families that are surface dependent. Family richness is the total number of families, and Order diversity is the calculated Shannon index for order. Tolerance is the weighed sum of families, with weights taken from the listed tolerance values (see Appendix and Fig 2).

<table>
<thead>
<tr>
<th>Site</th>
<th>Phys-Chem score</th>
<th>Urban %</th>
<th>Agro %</th>
<th>EPT Y/N</th>
<th>Surf Dep Y/N</th>
<th>Tolerance Y/N</th>
<th>Family richness</th>
<th>Order diversity</th>
<th># Macro-invertebrates collected</th>
</tr>
</thead>
<tbody>
<tr>
<td>Putnam Creek</td>
<td>18.0</td>
<td>2.7</td>
<td>22.7</td>
<td>5</td>
<td>6</td>
<td>100.3</td>
<td>19</td>
<td>1.70</td>
<td>124</td>
</tr>
<tr>
<td>County Drain No. 7</td>
<td>17.5</td>
<td>2.8</td>
<td>42.7</td>
<td>3</td>
<td>5</td>
<td>79.1</td>
<td>16</td>
<td>1.93</td>
<td>54</td>
</tr>
<tr>
<td>No Name Creek</td>
<td>10.4</td>
<td>10.5</td>
<td>27.6</td>
<td>2</td>
<td>8</td>
<td>101.3</td>
<td>18</td>
<td>1.49</td>
<td>154</td>
</tr>
</tbody>
</table>
DISCUSSION

High-quality aquatic systems provide many valuable ecosystem services such as clean water, habitat for fisheries, and recreation (Barbour, 2010). When strategizing how to manage aquatic systems, it is necessary to address how to determine their comparative quality. This study assessed habitat quality and overall stream health by creating stream profiles. These profiles used quantitative methods to characterize the plant communities, flow regime, and diversity of the macro-invertebrate communities within three streams.

A widespread method of assessing streams and rivers is the use of benthic macroinvertebrates, particularly aquatic insects, as indicators of pollution. The use of benthic macroinvertebrates is the basis for most aquatic biomonitoring programs currently in use (Dinakaran 2007). The benthic macroinvertebrates commonly analyzed when assessing streams include the orders Ephemeroptera, Plecoptera, and Trichoptera, (EPT). These three aquatic insect orders are particularly sensitive to water temperature, contaminants, and low dissolved oxygen. For these reasons, EPT are common bioindicators of freshwater biotic integrity (Pecher 2010). Biological communities provide an integrated response to pollutants and human disturbance within watersheds through their continuous exposure to the magnitude, duration and frequency of stressors (Barbour 2010). Given their habitat requirements and sensitivity to disturbance, noting the abundance and diversity of sensitive species, such as EPT, provides an avenue to assess ecosystem health within streams and rivers.

Results from our study showed that No Name Creek was home to 154 collected macroinvertebrate specimens, compared to Putnam Creek, which held 124, and County Drain No. 7, which held 54. The abundance of individuals in Putnam Creek did not correspond to the species richness of the stream overall, suggesting that this stream is dominated by a few
select species. The order and family diversity was also lowest in No Name Creek. Furthermore, the species that were present in No Name Creek are believed to be more tolerant of environmental pollutants and are not considered particularly sensitive species. Putnam Creek also showed a high tolerance number and low species diversity. Contrarily, although County Drain No. 7 had the least number of individuals, it had the greatest insect family and order diversity. County Drain No. 7 also had the overall lowest tolerance rating of all three sites. County Drain No. 7 held more species with low tolerance to contaminants which require high levels of dissolved oxygen. County Drain No. 7 also showed the greatest percentage of dissolved oxygen of the three streams sampled, and lowest conductivity. Thus it is prime habitat for sensitive macro-invertebrate species. County Drain No. 7 was in essence a healthy stream, while No Name Creek could be considered comparatively unhealthy.

There has been a rapid decline in biodiversity of global freshwater ecosystems (Pecher 2010). This is primarily in response to the vast detrimental effects caused by urbanization and other human activities. Factors such as watercourse and floodplain alteration, the related enhanced erosion, industrial waste, and sewage waste can lead to the loss of diversity in aquatic systems and homogenization of the ecosystem (Pecher 2010). Other potential factors causing the decline of biodiversity in freshwater ecosystems relate to the destruction of the riparian community structure. Physical conditions of streams, aside from water chemistry, form the most central aspects of the habitat in which lotic organisms evolve, adapt and interact. The assemblages that form under these physical conditions are precisely the factors that biological monitoring uses for assessment of stream condition (Frappier 2007).

Assessment of floodplain alteration and flow regime were addressed in our habitat scores. No Name Creek received the lowest score, 10.4, a ‘marginal’ score. No Name Creek showed the highest percentage of urban land usage, which caused blatant degradation of the landscape. Contaminants from motor vehicles, homes, and shops, along with local government decisions to reroute the stream and straighten its canals have presumably had profound negative effects on the ecology of No Name Creek. Putnam Creek received the highest habitat score, 18, upon evaluation. County Drain No. 7 was also categorized as suboptimal, but received a slightly lower habitat score of 17.5.

Loss of biodiversity may also be correlated with disturbances of flow regime. Flow regimes are considered a key driver of the ecology of rivers and their associated floodplain wetlands. Flow regimes directly influence the morphological processes, renewal of habitats, and the ecological richness of streams. Flow regimes can be modified by human use, such as water for drinking, irrigation, and hydroelectricity (Baran 2008). No Name Creek had the lowest water temperature and the greatest conductivity reading. Although the dissolved oxygen percentage in No Name Creek was reasonable, our data suggest that No Name Creek was the most degraded site. High conductivity (above 500 µS) in a stream is a cause for alarm, and conductivity readings for No Name Creek reached 712µS. The most likely cause for high conductivity in No Name Creek is pollution caused by anthropogenic factors such as agricultural run-off or poor waste management. Conductivity in Putnam Creek and County Drain No. 7 were similar and below the warning threshold of 500 µS.
Results from our study reflect the importance of accurately assessing the effect of humans on aquatic ecosystems by not only analyzing macroinvertebrates, but also flow regimes and riparian structure. Urbanization near No Name Creek had adverse effects on the overall productivity of the stream, specifically in terms of riparian structure, stream health, and macroinvertebrate biodiversity. Although Putnam Creek and County Drain No. 7 were challenged with urban and agricultural land use, both streams continue to provide sensitive species with quality habitat. This study shows that bio-assessments are necessary tools in order to propel conservation and focus restoration efforts within streams, which provide invaluable ecosystem services.

LITERATURE CITED


APPENDIX

Table 1. Physicochemical data measured at the three sites.

<table>
<thead>
<tr>
<th></th>
<th>No Name Creek</th>
<th>Putnam Creek</th>
<th>County Drain No. 7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riparian Width</td>
<td>10</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Riparian Structure</td>
<td>10</td>
<td>16</td>
<td>15</td>
</tr>
<tr>
<td>Bank Stability</td>
<td>12</td>
<td>20</td>
<td>20</td>
</tr>
<tr>
<td>Shading</td>
<td>5</td>
<td>17</td>
<td>20</td>
</tr>
<tr>
<td>Channel Alteration</td>
<td>5</td>
<td>17</td>
<td>19</td>
</tr>
<tr>
<td>Embeddedness</td>
<td>10</td>
<td>17</td>
<td>15</td>
</tr>
<tr>
<td>Benthic Silt Cover</td>
<td>15</td>
<td>19</td>
<td>16</td>
</tr>
<tr>
<td>Water Appearance</td>
<td>16</td>
<td>20</td>
<td>20</td>
</tr>
<tr>
<td>Averaged Total</td>
<td>10.4</td>
<td>18.0</td>
<td>17.5</td>
</tr>
<tr>
<td>Water Temperature (°C)</td>
<td>12.1</td>
<td>13.41</td>
<td>12.8</td>
</tr>
<tr>
<td>Conductivity (μS/cm)</td>
<td>712</td>
<td>497</td>
<td>406</td>
</tr>
<tr>
<td>Dissolved Oxygen (mg/L)</td>
<td>8.4</td>
<td>8.06</td>
<td>8.67</td>
</tr>
<tr>
<td>DO %</td>
<td>78.1</td>
<td>77.4</td>
<td>80.6</td>
</tr>
<tr>
<td>Erosion Habitat Type</td>
<td>Rifle, Run</td>
<td>Run</td>
<td>Run</td>
</tr>
<tr>
<td>Deposition Habitat Type</td>
<td>Lateral edge</td>
<td>Lateral edge</td>
<td>Lateral edge</td>
</tr>
<tr>
<td>Substrate Type</td>
<td>Gravel, Sand, FPOM</td>
<td>Gravel, Sand, Cobble</td>
<td>Gravel, Sand, Wood, CPOM</td>
</tr>
</tbody>
</table>

Figure 1. Abundances of invertebrate families found in each stream. *Hydropsychidae* were disproportionately abundant in No Name Creek, spiking at 75 individuals counted.
BONE ASSEMBLAGES IN THE E. S. GEORGE RESERVE

KATE ZEMENICK, CINDY BICK, RACHEL CABLE, DAVID GONTHIER, LAUREN MALONEY

Bone assemblages assist in the interpretation of mammal diversity in a given area and help estimate population densities. A recent study performed over forty years in the Amboseli ecosystem of Kenya compared recent bone assemblages with the current live community and found that not only can such assemblages indicate which species are present at the time, but they can also predict their relative abundances and community structure. To determine whether a survey such as this could be performed in the E.S. George Reserve, we sampled three habitats for evidence of mammals. We included evidence that would potentially translate to the fossil record, specifically: bones, burrows, scat, tracks, and chewing on bones. We found at least two bones in every site, and evidence of all other elements in each site, with the exception of scat. Although we cannot infer population densities from such a low sample size, this study provides groundwork that can be built upon to learn more about the bone assemblages at the ESGR.

INTRODUCTION

Bones are important to our understanding of vertebrates not only in life, but also long after death. The study of recent bone assemblages is vital because bones do not decompose as the rest of the body does, and they can more easily translate into the fossil record (Behrensmeyer 1984). We can also deduce the mechanisms of death and mortality rates within a population from bone assemblages (Vrba 1980), as well as follow the process of nutrient cycling back into the environment from once-living animals (C. Badgely, pers. comm.). Over long periods of time, the observation of bone assemblages can also indicate changes in habitat as the species and abundance of animals change in a certain area (Cutler et al. 1999, Behrensmeyer et al. 1979). The approximate time since the death of the animal can also be quickly determined, given the environmental conditions, by the weathering stage of the bone (Behrensmeyer 1978).

The study of the process bones undergo from deposition in the environment to incorporation into the fossil record and excavation is termed taphonomy, which has progressively become a method for answering ecological questions about past ecosystems and even how they relate to modern ecosystems (Olson 1980). A recent study performed over forty years in the Amboseli ecosystem of Kenya compared recent bone assemblages with the current live community and found that not only can such assemblages indicate which species are present at the time, but they can also predict their relative abundances and community structure (Western and Behrensmeyer, 2009). Since these bone assemblages are what is most often found in the fossil record, Western and Behrensmeyer
inferred that the fossil record in such areas would accurately describe the animal population and community structure at that time.

To determine whether it is feasible for a similar survey to that performed in Amboseli to be performed in the E.S. George Reserve, we decided to sample three habitats within the reserve for evidence of mammals. We included evidence that would potentially translate to the fossil record, specifically: bones, burrows, scat, tracks, and chewing on bones. The George Reserve is currently known to house 42 species of mammals. These include small burrowing mammals such as the white-footed mouse \((Peromyscus maniculatus bairdii)\), and the eastern chipmunk \((Tamias striatus)\), which leave distinct parallel gouges in bone when they sharpen their teeth. Medium-sized burrowers such as least weasels \((Mustela nivalis)\) and skunks \((Mephitis mephitis)\) and large burrowers such as woodchucks \((Marmota monax)\) also reside in the reserve. Larger meso-carnivores such as red foxes \((Vulpes vulpes)\), gray foxes \((Urocyon cinereoargenteus)\), and coyotes \((Canis latrans)\) inhabit the George Reserve, and their presence can be noted by the movement of large bones and large gnaw marks on bones. Medium-sized omnivores such as raccoons \((Procyon lotor lotor)\) on the reserve leave tracks and scat containing seeds and hair. The largest of the mammals found on the reserve, the white-tailed deer \((Odocoileus virginianus)\), leaves distinctive scat and tracks (C. Badgely, pers. comm.).

While these animals inhabit different habitats within the reserve, those different habitats also lend to easier sighting of evidence of mammals. Forested areas with sandier soil that supports brushy undergrowth less than mucky soil, fields with less leaf litter than forests, and pine stands with more acidic soils and less large leaf litter than mixed woods, all have low ground cover and more open spaces for sighting of mammal evidence on the ground (C. Badgely, pers. comm.).

**MATERIALS AND METHODS**

The surveys of vertebrate presence were conducted at 3 different sites on the Edwin S. George Reserve in Pinckney, MI on October 3rd, 2010. The surveys were done at the Blowout Area (transect oriented N 25°E from S end to N end), near Evan’s Old Field (oriented due E, beginning at the west end and heading east), and the Red Pine Plantation (oriented due N in NW West woods where the fence has a corner). With the exception of the third site (red pine plantation), our survey areas were 200 meters long with 30 meters perpendicular on each side. Surveyors walked back and forth in a straight line perpendicular to the 200 meter transect and parallel to the 30 meter transect. Site 3 was 100 meters long with 30 meters perpendicular on each side.

Elements such as bones, chewing marks on bone, burrows, trackways, and scat were indications of vertebrate presence. Metal pokers were used to tamp down on the soil and the leaf litter, as the feel of metal striking bone material is easily identifiable. The location of each element was marked with a flag. The bones were collected and information on the taxon, age, and weathering stage were recorded. Chewing marks on bones were identified to species if possible. Burrows were measured to the nearest centimeter and recorded. We sorted the burrows into different size classes to determine
possible inhabitants (Table 1). Scat was sorted to detect animal hair, bones and other identifiable characteristics of an animal’s diet.

<table>
<thead>
<tr>
<th>Table 1: Burrow Sizes and Possible Inhabitants</th>
</tr>
</thead>
<tbody>
<tr>
<td>Size (cm)</td>
</tr>
<tr>
<td>&lt;3</td>
</tr>
<tr>
<td>3-10</td>
</tr>
<tr>
<td>10-25</td>
</tr>
<tr>
<td>25-30</td>
</tr>
</tbody>
</table>

Estimations of ground cover vegetation and leaf litter cover were also recorded at all three sites. At each site, the Minimum Number of Individuals (MNI) and the Minimum Number of Species (MNS) was determined. MNI was number of different species found using only bones as evidence. MNS was inferred by all elements we recorded at each site, including bones (Western and Behrensmeyer, 2009).

RESULTS

We found at least two bones in every site, and evidence of all other elements in each site, with the exception that we did not find scat in Site 3.

Description of Bones - MNI

In Site 1 we found a deer skull and various parts of a raccoon (*Procyon lotor lotor*) skeleton (Table 2). The deer skull was determined to be an adult, and in weathering stage (WS) 0. There was a bit of flesh still attached, and chewing was present on the back of the skull and below the right eye. The teeth marks resembled a meso-carnivore, either a gray fox (*Urocyon cinereoargenteus*), red fox (*Vulpes vulpes*) or coyote (*Canis latrans*). The raccoon specimen was determined to be in WS 0 because the skull was in fragments but the bone did not show signs of cracking. The sutures were not quite complete, the Humerus ends were not fused, and the 3rd molar was just erupting, so it was determined to be a juvenile at time of death. The MNI at Site 1 is therefore 2.

Three different deer bones were found in Site 2 (Table 2). The lumbar vertebrae and tibia were both in WS 0 and juveniles, so it is possible they are from the same individual. The former was found on a fallen tree and the latter under a pile of leaf litter. The calcaneum were in WS1 and chewed by a meso-carnivore. Although 3 different bones were found, they were all the same species, making MNI = 1.

In Site 3, three different bones from three different species were found (Table 2). The woodchuck (*Marmota monax*) Humerus was found at the base of an oak. It was an adult in WS. The adult gray fox cranium was in WS 1+. Lastly, an adult raccoon cranium was found in WS 1. The MNI at Site 3 = 3.

| Table 2: Description of bones found at each site, with notes about carnivore chewing so that other species in the area can be inferred. R = right, L = left. |
Summary of Mammal Evidence – MNS

The total evidence for presence of species in each site is shown in Table 3. The species found at all sites included: white tailed deer, raccoon, chipmunks, mouse/vole, woodchuck, skunk/weasel, rodents, a gray fox, meso-carnivore (coyote or fox) and an unidentified omnivore. Not all species were present in each site. Due to overlap or inconclusive evidence, the minimum number of species (MNS) at each site was much lower than the total observations of species.

Site 1 MNS = 5, Site 2 MNS = 5 and Site 3 MNS = 6 (Figure 1). The MNS in site 3 was not 8 because the rodent chewing may have been a rodent burrowing in the site, and the fox or coyote chewing could have been due to a gray fox, already accounted for with bone evidence. Although the gray fox skull was at a greater WS than the other two bones that had meso-carnivorous chewing marks, the WS may not always accurately reflect time since death.

The Red Pine Plantation (Site 3) tends to have a higher density of MNI and MNS than the Oak Savanna or Oak-History Forest (Figure 1). Each element was present at Sites 1 and

### Table 3

<table>
<thead>
<tr>
<th>Site</th>
<th>Species</th>
<th>Bones Present</th>
<th>Notes</th>
</tr>
</thead>
</table>
| 1    | Odocoileus virginianus | Cranium, P2-M3 | Pre-maxilla missing  
Flesh attached  
Meso-carnivore chewing |
| 1    | Procyon lotor lotor | RL maxilla with RL P3 and P4, R M2 erupting.  
Pelt present with pre-maxilla and incisors  
Lower jaw: L: D11, D12, DC,C, P3, P4, M12(erupting)  
Lower jaw R: D11, DC,C, P2, P3, PF, M1  
Vertebrae: HTL, CERV4-CAUD3  
Ribs: 4 articulated, 6 additional.  
RL capula, humerus, femur  
R: tibia, fibula, complete hind foot  
L: ulna, meta-podial |
| 1    | Odocoileus virginianus | Lumbar vertebrae | Unidentifiable chewing on lateral processes |
| 2    | Odocoileus virginianus | L tibia | Unidentifiable chewing on proximal end |
| 2    | Odocoileus virginianus | RL calcaneum | Meso-carnivore chewing |
| 3    | Marmota monax | L humerus | Meso-carnivore chewing on proximal and distal edges |
| 3    | Urocyon cinereoargenteus | Cranium | Rodent and meso-carnivore chewing on ventral side |
| 3    | Procyon lotor lotor | Cranium | Rodent chewing on cranium and palate |
2, however Site 3 lacked trackways. However, of the other 4 elements Site 3 tended to have a much higher density (Figure 2).

Table 3: Elements Found at Each Site

<table>
<thead>
<tr>
<th>site</th>
<th>species</th>
<th>element</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>deer</td>
<td>bone, 6 scat, 4 track</td>
</tr>
<tr>
<td>1</td>
<td>raccoon</td>
<td>bone, 3 scat</td>
</tr>
<tr>
<td>1</td>
<td>fox/coyote</td>
<td>chewing</td>
</tr>
<tr>
<td>1</td>
<td>chipmunk</td>
<td>15 burrows</td>
</tr>
<tr>
<td>1</td>
<td>omnivore</td>
<td>1 scat</td>
</tr>
<tr>
<td>2</td>
<td>deer</td>
<td>3 bones, 4 scat, 1 track</td>
</tr>
<tr>
<td>2</td>
<td>raccoon</td>
<td>1 scat</td>
</tr>
<tr>
<td>2</td>
<td>fox/coyote</td>
<td>chewing</td>
</tr>
<tr>
<td>2</td>
<td>chipmunk</td>
<td>20 burrows</td>
</tr>
<tr>
<td>2</td>
<td>mouse/vole</td>
<td>4 burrows</td>
</tr>
<tr>
<td>3</td>
<td>gray fox</td>
<td>bone</td>
</tr>
<tr>
<td>3</td>
<td>woodchuck</td>
<td>bone</td>
</tr>
<tr>
<td>3</td>
<td>raccoon</td>
<td>bone, 11 scat</td>
</tr>
<tr>
<td>3</td>
<td>skunk/weasel</td>
<td>3 burrows</td>
</tr>
<tr>
<td>3</td>
<td>chipmunk</td>
<td>13 burrows</td>
</tr>
<tr>
<td>3</td>
<td>mouse/vole</td>
<td>9 burrows</td>
</tr>
<tr>
<td>3</td>
<td>rodent</td>
<td>3 chewing</td>
</tr>
<tr>
<td>3</td>
<td>fox/coyote</td>
<td>bone, chewing</td>
</tr>
</tbody>
</table>

Figure 1: MNI and MNS per Unit Area. The density of Minimum Number of Individuals (MNI) indicated by bones, and the Minimum Number of Species (MNS) indicated by all elements at each site.

Figure 2: Element per Unit Area. This graph shows the density of each element (tracks, scat, bones, burrows, and chewing) per Unit Area at each site.
DISCUSSION

Bone assemblages assist in the interpretation of mammal diversity in a given area and help estimate population densities. Accurate reconstructions of ancient community ecology depend on how closely fossil assemblages match species richness and relative abundance in the original living communities (Western and Behrensmeyer 2009). Unfortunately, our bone assemblages were not comprehensive enough to predict population densities due to the low number of bones found at each site and the lack of living population densities. However, we were able to estimate the minimum number of species (MNS) and the minimum number of individuals (MNI) within each transect.

For site 1 we estimated the MNS as 5, for site 2 we estimated the MNS as 5 and for site 3 we estimated the MNS as 6. Although we find solid evidence of numerous mammals on each site, it is difficult to estimate how many mammals were actually present. Minimum number of species is a conservative judgment for the number of mammals present. MNS leads you to assume that scat present, no matter the amount, only indicates 1 individual. Similarly, more than one burrow in one area also indicates only 1 species when in fact it could be more than one or perhaps even another species; It is difficult to distinguish between a mouse and a vole burrow without further evidence of either species.

The minimum number of individuals (MNI) estimate is even more conservative because it is based off the species identified by bones and considers bones found from the same species at the same age to be the same individual. For site 1 we found a MNI of 2, for site 2 we found an MNI of 1 and for site 3 we found an MNI of 3. Consequently the largest numbers of MNI and MNS we found at site 3. This may be due to either an increase in population density or because it is easier to find bones, scat, and burrows in the forest floor of a pine plantation. The understory vegetation and the leaf litter of site 2 were very
dense and could have impeded bone discovery. Furthermore, when the group surveyed site 3 we had developed a search image for bones, scat and burrows. It is possible that we were able to find them more easily, thereby increasing the frequency of finds for that site.

Although the study did not allow us to estimate species population density it did reveal existence of species within the reserve. We found bones from raccoon, deer, woodchuck and grey fox. The grey fox is significant because little is known about their presence within the George Reserve. At all three sites, we saw meso-carnivore chewing on some of the bones, suggesting the presence of coyote or fox. We can also hypothesize about the placement of the bones and how they came to rest in that particular space. For example, the deer vertebra from Site 2 was lodged into a fallen log, suggesting another animal carried it there - perhaps a raccoon. Raccoons could also be the culprit in the placement of the two skulls found under trees in Site 3, as raccoons are carnivores and are avid tree climbers. Site 3 also had a high frequency of raccoon scat.

Further research could compare more transects in different habitats in the George Reserve and then compare the range of species in each habitat. With further research the enigmas could be answered; are certain environments easier to find traces of mammals, or are they in fact more trafficked by mammals?

ACKNOWLEDGEMENTS

We would like to thank Catherine Badgley for her inspiration and expertise in mammal biology

WORKS CITED


London.


BLACK/RED OAK REGENERATION STRATEGIES: DO SEEDLINGS INTELLIGENTLY ALLOCATE BIOMASS IN RESPONSE TO STRESS?

RAFAEL D’ANDREA, CINDY BICK, SAHAR HAGHIGHAT, ABIAGIL HYDUKE, LAUREN MALONEY, AND NICK REO

ABSTRACT

Oaks are common and important trees in North American temperate forests, which are however declining due to low self-replacement rates. Shade and browsing restrain the shoot’s growth, but do not prevent accumulation of root mass. Such mismatch between below and above ground parts introduces the interesting possibility of unequal biomass allocation as a strategy for outcompeting rivals to the canopy. Here, we investigate whether seedlings respond to environmental stress. Our results show that biomass allocation bias is indeed stronger under harsher conditions, supporting the idea that such bias is adaptive.

INTRODUCTION

Quercus is a prevalent genus of North American hardwood trees. Not only are oak widespread, but they also contribute to a wide range of both biological and economic uses. Quercus provides an ecosystem function by supplying food and shelter for wildlife, contribute to the biodiversity of the forest, and from an anthropocentric perspective they have high economic value. They are used to make many wood-based products including pallets, furniture, and building materials. In terms of quality and quantity of saw timber, white oak is arguably North America’s most valuable hardwood species (Abrams 2003). Although oaks are a symbol of American forests and have a high economic value, oak forests are in decline. Oaks are gradually relinquishing their long-term historical dominance in forests of eastern North America (Lorimer 2003).

Oak species have numerous factors working against their regeneration. Obstacles include competitive influences of faster-growing species, production of quality seedling stock, and of particular importance, the influence of whitetail deer herbivory (Oswalt et al. 2006). Oaks form a primary successional genus, and due to their shade intolerance it is difficult for oaks to regenerate in their own shade. Animal browsing, especially by deer, is also detrimental to young oak seedlings and inhibits their growth. Quercus regeneration requires an edge environment; one that is more mesic than exposed, open sites, but less competitive than the deep shade of a forest understory (Crow 1988).

Oak seedlings are able to resprout should the above ground stem collapse under disturbance or unrelenting stress. This is facilitated by the development of large root shoots (Johnson 1994). Young plants put energy into their roots to build up a root system that can compensate if the above ground seedling is killed. Eventually, the plant is able to send up another shoot from its...
root mass. This underground storage can be seen as a proactive defense mechanism that can help it eventually overcome obstacles to its growth, and is made possible by extra allocation of biomass to the root as opposed to the aboveground stem. This dieback-resprout process can happen many times before the seedling finally reaches safety at some threshold height. The tree then undergoes a series of resprouting cycles before reaching adult size.

In this study, we investigate if and how this cycle differs across different growing conditions. We collected samples from two plots in the E. S. George Reserve which presumably present different levels of stress to seedling growth. Our basic expectation is that seedlings would undergo a greater number of cycles in the more stressful environment. If that is true, then the following consequences should follow:

- Seedlings should be taller in the less stressful environment.
- Root should be older than stem in both sites, and the age difference should be higher in the most stressful environment.
- Correlation between above ground and below ground measures should be higher in the less stressful environment.
- If biomass allocation is disproportionately higher for below ground parts, such disparity should be stronger in the more stressful environment.

We chose an oak forest understory and a semi-open field as our experimental plots, and assumed the two main sources of stress there should be deer browsing and shade. Light is obviously more of an issue in the understory than in the field. Additionally, we propose that deer browsing should be more intense in the forest, as deer probably spend more time in sites where they feel safer from predators, though we do not have data to confirm this idea. Therefore, we expect that the field should represent the less stressful environment, and the aforementioned predictions should hold correspondingly.

MATERIALS AND METHODS

Black and red oak (*Q. velutina, Q. rubra*) seedlings\(^1\) were collected from an open field and a forested area in the E. S. George Reserve on October 9, 2010. The above ground stem and below ground root were collected from seedlings approx. 40m apart within a 400m transect. Seedling height was taken in situ, from samples ranging from 40cm to 120cm. “Cookies” were cut from below and above ground portions as close to the root collar as possible. Calipers were used to measure the diameter of the cookies, and the annual rings were then counted under microscopes to age the stem and root of each seedling.

RESULTS

Fig 1 confirms that our two plots differ significantly in terms of stress to young trees. Seedlings in the field plot are significantly taller than those in the woods. Fig 2 shows higher correlation between above ground (AG) and below ground (BG) diameter in the field, and Fig 3 indicates

\(^1\) As seedlings from these two species are very similar, and as hybrids are common in the site of collection, we did not try to distinguish between them.
higher variance in the ratio (Height)/(BG diameter) in the woods. Both imply higher variability and lower predictability in above ground measures for a given value of BG diameter in the woods compared to the field.

Figure 1. Distribution of heights in increasing order for the field and woods plots. The mean(SE) for woods and field are 53.8(7.3)cm and 70.4(15.4)cm, respectively. Two-sample independent t-test gives p=0.006.

Figure 2. Regression of above ground diameter on below ground diameter for our two plots.

Figure 3. Deviations from average (Height)/(BG diameter) for woods and field.

Fig 4 shows the average age of root and stem in both sites. We see that even though BG parts are the same age across plots, the AG part is significantly younger on average in the woods, indicating more dieback-resprout cycles. Likewise, the average ratio (AG age)/(BG age) is significantly lower in the woods than in the field (Fig 5). The same plot can be made using diameter in lieu of age, showing similar results (Fig 6).
As for biomass allocation, we define the quantity \((\text{diameter})^2/\text{age}\), here called relative growth rate (RGR), as a surrogate for biomass input rate\(^2\). Table 1 displays mean RGR for root and stem across sites.

Table 1. Data on biomass allocation. The numbers across columns are significantly different in the \(\alpha = 0.1\), as shown by the p-values.

<table>
<thead>
<tr>
<th></th>
<th>Mean RGR (cm(^2)/yr)</th>
<th>Field</th>
<th>Woods</th>
<th>F</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>BG</td>
<td>42.3</td>
<td>26.8</td>
<td>3.3</td>
<td>0.09</td>
<td></td>
</tr>
<tr>
<td>AG</td>
<td>29.2</td>
<td>13.5</td>
<td>3.3</td>
<td>0.09</td>
<td></td>
</tr>
</tbody>
</table>

\(^2\) By doing this, we are assuming that biomass scales with the square of the diameter. The justification for this is not clear from first principles, but it seems to find empirical validation in the literature (D’Andrea 2010)
As expected, biomass input rate is generally higher in the field. The reason is clear: there is higher availability of carbon there. Also, in both sites more biomass is invested in the more durable below ground parts. More importantly, this data indicate that such bias toward the underground parts is more pronounced in the harsher environment. This can be seen by dividing the first row by the second: we have a ratio $\frac{\text{RGR}_{\text{BG}}}{\text{RGR}_{\text{AG}}}$ of 1.4 for the field and 1.99 for the woods. Additionally, comparison of the same part of the plant across the two sites shows a higher discrepancy in biomass allocation for the AG part: dividing the Field column by the Woods column, we get 1.58 for BG and 2.16 for AG.

**DISCUSSION**

Our data confirm all our expectations. The forested plot proved a more challenging environment to the seedlings, which then underwent a greater number of dieback-resprout cycles. For that we observe: a greater difference in age between above and below ground parts, a greater variability and consequently less predictability in height as a function of below ground diameter or age, and preferential biomass allocation to the root versus the above ground stem.

Moreover, our results show that the difference between biomass stored in the root and the stem is above and beyond what would be expected from the constant replacement of the stem. By normalizing biomass (represented by $\text{diameter}^2$) with respect to age, we obtain a rate of annual biomass input, which, upon comparison between root and stem, should show no difference if the null hypothesis of equitable allocation were true. This implies that the plant seems to respond to stress by differentially allocating biomass to below ground parts. This plasticity might be adaptive when a seedling is faced with an environment in which survival and growth prove difficult. In such a scenario, it is plausible to assume that the seedlings will allocate more to underground storage, rather than squandering its resources on aboveground biomass, which might not survive stress caused by conditions such as herbivory or forest fires. Furthermore, this growth pattern could facilitate better competitive ability against other seedlings by allowing use of stored below ground biomass to rapidly grow and outcompete surrounding seedlings.

A consequence of such cycles is the presence of seemingly young seedlings possessing truncated stems that are disproportional to their large root collars, which serve as better indicators of their true age.

**CONCLUSIONS**

This study contemplates the idea that bias toward underground biomass allocation by plants might be the result of adaptive evolution against precarious environments (Dickson and Nelson 1982; Kays and Canham 1991; Nelson and Dickson 1981). This interesting possibility should be investigated in other genera of plants. Primary successional plants are expected to show stronger bias, as those are the ones under stronger selection due to their lower tolerance to stress (Canham et.al. 1999) If proven general, we propose that this mechanism may contribute
to coexistence between plants typical of different succession stages, as it provides a tradeoff between competitive ability in different conditions. Such tradeoff should be especially relevant in stochastic, unstable environments.

In more practical terms, further research in the relationship of the above and below ground age and rate of growth could assist in the development of successful regeneration strategy for forest managers and those concerned with the future of one of North America’s quintessential trees.

**LITERATURE CITED**


LEAF TRICHOMES, ACARODOMATIA, AND MITE ABUNDANCE ON TEMPERATE WOODY PLANTS

DAVID GONTHIER, RACHEL CABLE, AND RAFAEL D’ANDREA

ABSTRACT

Mites are important members of arboreal leaf arthropod communities. Some species appear to be important in controlling plant pathogens. The presence of acarodomatia, including tuft domatia made of densely packed leaf hairs, on leaves has been shown to increase the abundance of mites on leaves. Yet few consider if leaf hairiness in general could also be important to mite abundance on leaves. We investigated this hypothesis by comparing the abundance of mites across woody species that were hairless (smooth leaved), hairy leaved, or had acarodomatia present. The results of our study suggest plants with acarodomatia host more mites than smooth leaved species, but did not differ in mite abundance with species that were hairy leaved. There was no difference in mite abundance between smooth and hairy leaved species. We also observed a positive relationship between the number of mites per leaf and the number of domatia per leaf within those species that harbored domatia. These data did not find strong evidence that leaf hairiness alone is important to mite abundance. However, they suggest acarodomatia do indeed improve the quality of leaf habitat for mites and therefore adds to the underrepresented literature on mite ecology on plants.

INTRODUCTION


It comes as no surprise that the presence of acarodomatia on plants typically results in a greater abundance of mites on leaves (Walter & O’Dowd 1992, O’Dowd & Willson 1991, Willson 1991). Previous research has suggested that shaving off of tuft domatia (made of densely clustered leaf hairs) reduces mite abundance on leaves (Walter & O’Dowd 1992). Further, generally species that have leaf domatia have more mites per leaf than species without domatia (Walter & O’Dowd 1992). Experimental studies suggest that mites benefit from domatia by avoidance of predators (Norton et al. 2001). Yet because a large proportion of the acarodomatia found on plants are tuft domatia (in the temperate region), it could be that it is leaf hairs (leaf
trichomes) in general that provide a resource to mites on leaves. Previous studies did not consider this level of comparison (Walter & O’Dowd 1992), although it has been considered (Walter 1996). Hairy leaved species might provide an intermediate leaf habitat for mites, whereas hairless or smooth leaved species provide a low quality habitat and domatia hosting plants provide high quality habitat.

The purpose of this study was to test if woody plants with acarodomatia would host greater mite abundance than hairy or smooth leaved species. Additionally, we investigated if plants of hairy leaved species host more mites than smooth leaved species. Finally, we tested the hypothesis that if acarodomatia are a resource to mites than leaves with more domatia should have more mites.

METHODS

We conducted this study at the E. S. George reserve in Eastern Michigan in October of 2010. First, we selected plant species with smooth leaves, hairy leaves, and acarodomatia from species previously surveyed for acarodomatia in Gonthier et al. (2010) These species included smooth leaved species: big-tooth aspen (*Populus grandidentata*), sassafras (*Sassafras albidum*), white oak (*Quercus alba*), sumac (*Rhus* sp.); Hairy-leaved species: Virginia creeper (*Parthenocissus quinefolia*), black raspberry (*Rubus occidentalis*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginiana*); Domatia baring species: riverbank grape (*Vitis riparia*), black cherry (*Prunus serotina*), red maple (*Acer rubrum*), red oak (*Quercus rubra*), American elm (*Ulmus americana*), pignut hickory (*Carya glabra*). Next, we collected green leaved samples (3 per individual) from 3 individuals of a given species. We then surveyed an 8-cm² quadrant along the mid-vein of every leaf. To capture the most important habitat for mites within leaves the segment sampled were located starting at the base of the vein near the petiole and was four cm long by one cm on in each direction of the mid-vein. On each leaf sampled we confirmed the smooth, hairy, or domatia baring nature of the leaf and then counted the number of domatia present, the number of mites, and number of mites within domatia within the per 8-cm² quadrant. Mites were assigned to morpho-species based on color and distinguishing marks, however species resolution appeared low and species level comparisons were not considered here.

To determine the relationship between leaf characters and mite abundance a number of comparisons were made. First, we compared the average number of mites per 8-cm² quadrant (per leaf hereafter) on a leaf across all species (9 leaves per species) using a one-way analysis of variance (ANOVA) with Tukey’s pairwise comparisons. Next, we grouped species that fell within the categories smooth, hairy, and domatia leaved species and compared the mean number of mites per leaf across groups with one-way ANOVA with Tukey’s pairwise comparisons. Finally, within plants with acarodomatia, we compared the number of mites per leaf with number of domatia per leaf using non-parametric Spearman’s Rank Correlation ($r_s$). Similarly, we found the relationship between the number of mites per leaf within acarodomatia and the number of
domatia per leaf with Spearman’s Correlation. The number of mites per leaf deviated from normality therefore we natural log transformed (ln[mites + 0.001]) this variable for all tests except for the Spearman’s correlations.

RESULTS

The raw count per species can be seen in Fig 1. Each bar corresponds to an average over 9 leaves. There were significant differences in mite abundance between some species (F = 3.8, P =0.001). American elm had greater mite abundance than aspen (P = 0.004) and Virginia creeper (P = 0.042). Dogwood (P = 0.014) and red oak (P = 0.029) also had greater mite abundance than aspen. There were no other pairwise differences between species.

Figure1. Mite abundance per leaf for each of the 14 species observed. Different letters indicated significant differences indicated by Tukey’s pairwise comparisons.

When species were clumped into the three categories domatia baring, hairy leaved, and smooth leaved species there were differences in mite abundance across groups (Fig. 2, F = 11.89, P = 0.001). Tukey’s pairwise comparisons of the natural log transformed data indicate domatia baring species had greater number of mites per leaf area sampled than smooth leaved species (P = 0.005), however they did not differ from hairy leaved species (P = 0.09). Further smooth leaved species did not differ from hairy leaved species (P = 0.5).
When only woody plants that contained acarodmatia were considered, the importance of acarodmatia to mite abundance was elaborated. There was a positive relationship between the number of domatia on a leaf and the number of mites per leaf (Fig. 3, $r_s = 0.307$, $P = 0.024$). Further, the number of mites found within domatia also increased with the number of domatia per leaf on plants that contained domatia (Fig. 4, $r_s = 0.406$, $P = 0.044$).
DISCUSSION

A significant difference in mite abundance between woody species with domatia on their leaves and those with no hairs on their leaves is immediately apparent from our study. However, the difference in mite abundance between species with domatia and those that are merely hairy without domatia was not statistically significant. This implies that mites do not preferentially choose to live on leaves of species with domatia. Instead, since all domatia species also have hair on their leaves or stems, mites preferentially inhabit species with hairy leaves in general, as observed by David Walter (1996). This conclusion changes, however, when Flowering Dogwood is considered a domatia species and not hairy, as it is in some classifications (Reed 2006), but not ours. With the new classification, there is a significant difference between the mite abundance on domatia-bearing species compared to both smooth and hairy leaf species (Fig. 5 F = 6, P = 0.005), thus mites significantly prefer domatia over all leaf morphologies. This difference remains when Flowering Dogwood is completely removed from the study.

Figure 5. Means of mite abundance per category with Flowering Dogwood categorized as domatia-bearing.
The presence of such an outlier that can be differentially classified confuses the outcome of our data, so we decided to exclusively examine the domatia-bearing species for actual mite usage of domatia. Our data shows that species and leaves with larger abundances of domatia had correspondingly larger abundances of mites. Considering the observed area of each leaf was kept constant throughout the study, this indicates that mites actually inhabit or use the domatia, or else the density of mites would simply correspond to the area of the leaf and not the number of domatia. We also noted that with rising domatia abundance in the set area, the mite abundance found within those domatia also rose. Thus, we have seen with our very eyes and statistically proven that mites are using the domatia on domatia-bearing leaves.

The total amount of mites we encountered was much lower than what was expected from previous studies (Pemberton & Turner 1989, Willson 1991, O’Dowd & Willson 1991), and our data was consequently easily influenced by the classification of Flowering Dogwood. The low overall abundance of mites might be due to the season in which we performed our study. By mid-October, many of the leaves had changed colors and fallen from our study species on the George Reserve. Although we attempted to only acquire and examine green, healthy leaves, sometimes the only available specimens were from trees already changing color and losing leaves. It is possible that the mites sense this change and abandon the leaves before they fall, leaving us with very few mites on our specimens. It would be recommended to repeat this study during the summer, when mite communities would be established on leaves.

In conclusion, we demonstrated that domatia plants were inhabitant by more mites than plants with smooth leaves. Hairy leaved species were intermediate between these two groups, however were strongly influenced by the outlying species flowering dogwood. This study provides overall support for the idea that acarodomatia provide some benefit to mites living on leaves. Yet further elaboration on the mechanism of those benefits is pending.

ACKNOWLEDGEMENTS

Thanks to Ivette Perfecto, John Vandermeer, and David Allen for help with project design and logistics.
LITERATURE CITED


INQUILINE ABUNDANCE IN RELATION TO EXTERNAL VARIABLES OF SARRACENIA PURPUREA OF HIDDEN LAKE BOG AT THE E.S. GEORGE RESERVE

ABIGAIL HYDUKE, BINBIN LI, LAUREN MALONEY AND KASSANDRA SEMRAU

ABSTRACT

The digestion of prey items in carnivorous plants such as Sarracenia purpurea is facilitated by a community of organisms which inhabit the water body collected by the modified leaf or pitcher. Our study was to determine specific variables which may correlate, positively or negatively, with abundance and diversity of the inquiline community. We extracted liquid and recorded external characteristics of select pitcher plants within each clump we examined. We counted larvae of Wyeomyia smithii (mosquito) and Metriocnemus knabi (midge) and number of prey items within each sample. We found a positive correlation between the mosquito and midge larvae, indicating their ability to cohabit. We found more larvae in pitchers with bigger openings and in pitchers of color class 4 only, suggesting that presence of inquiline constituents and prey items are subject to numerous variables.

INTRODUCTION

Sarracenia purpurea, the purple pitcher plant, is a carnivorous bog species whose key morphological feature is the modified leaf (the ‘pitcher’) that serves to capture and store rainwater. The pitcher provides a source of stagnant water for insect larvae and a variety of other species including water rotifers, paramecia, tetrahymena, and a diverse bacterial community (collectively an inquiline community) (Heard, 1994). Hairs located on the plant provide a slippery surface which causes insects to fall into the water-filled pitcher, effectively acting as a pitfall trap (Juniper et al, 1989, Newell & Nastase, 1998). A variety of factors prevent the insect from escaping and it eventually drowns, at which point the inquiline community begins to break down the dead prey.

The larvae of two species of insects are found only within S. purpurea, the midge Metriocnemus knabi and the mosquito Wyeomyia smithii (Heard, 1994). The larvae of M. knabi facilitate the survival of W. smithii larvae because they shred insect prey for W. smithii to then consume through filter feeding (Bradshaw, 1983, Heard, 1994). Both species exist in a mutualistic relationship with S. purpurea, whereby W. smithii and M. knabi aid S. purpurea in its digestion of insect prey and receive shelter and food sources in return. Numerous studies have examined the methodology by which S. purpurea attracts insects as prey items. It has been shown that ants are attracted to S. purpurea with higher sugar levels, implying that pitchers with a lot of extrafloral nectaries receive more visits by foraging insects (Bennett & Ellison, 2009). Additionally, S. purpurea coloration varies from green to red and extrafloral nectaries are associated with red coloration, so we would expect plants with red coloration to contain more prey items than their greener conspecifics (Bennett & Ellison, 2009). Often within clumps of S. purpurea it is likely to find a wide range of coloration among individuals.
While coloration does not seem to be important to insect prey, it may be that *W. smithii* females use color when deciding where to oviposit. Additionally, pitchers with larger peristome are expected to contain more prey items (Wolfe, 1981, Cresswell, 1993), be less likely to dry out (Kingsolver, 1979), and contain more *M. knabi* larvae (Paterson & Cameron, 1982, Heard, 1994). Furthermore, larger clumps of *S. purpurea* are likely to attract more insect prey due to the higher availability of nectar, and therefore provide a more nutrient rich environment for inquiline communities. In this study we are interested in factors affecting the abundance of *W. smithii* and *M. knabi* larvae in pitchers exhibiting a variety of characters. We will examine abundances of *W. smithii* and *M. knabi* larvae within pitchers as relating to the size of the peristome, the coloration of the pitcher, distance to the edge of the bog, and number of pitchers per clump. Additionally, we will compare the relative abundances of midge and mosquito larvae with one another and the number of dead prey items.

**METHODS**

We chose 4 sites on the boundaries of the Hidden Lake Bog and laid out a transect from the open water to the edge of the swamp. We selected 6 pitcher plant clump or communities within 5 meters of each side of the transect. Each transect was at least 20 meters apart to obtain a broader sampling size around the lake. We chose clumps of greater than five individual pitchers and created a color categories from 1 to 5. 1 being nearly 100 % green, 2 mostly green, 3 half green half red, 4 mostly red and 5 being nearly 100% red. We selected plants of each color level to extract the contents of the pitcher as long as we could fit a turkey baster into the peristome or opening of the pitcher plant and as long as there was sufficient liquid to extract. In addition to color class, we recorded size of clump, including width of the colony and number of pitchers and proximity to the open water. We also recorded peristome diameter of those pitchers from which we extracted liquid. We transported the liquid from the pitchers in vials back to the lab to count mosquito and midge larvae. We also recorded the number of dead insects in each pitcher. To analyze the correlation between pitcher plant variables and midge and mosquito larvae presence we used spss program Pearson or spearman.

**RESULTS**

The color of pitcher plant shows no significant correlation with the number of mosquito larvae ($R^2=0.002$, $p=0.875$) or midge larvae ($R^2=-0.004$, $p=0.975$). We pooled the data from Color 2 to Color 5 which displayed red veins while Color 1 was all green. Only the presence of midge larvae showed significant difference. They were more in pitcher plants which veins turned red ($Z=-1.967$, $p=0.049$). However, mosquito larvae and other dead things did not show the same trend.

Mosquito larvae and midge larvae displayed positive correlation with the proportion of pitcher plants with Color 4 in a clump (Fig.1). But no similar correlations were found in the other color categories. In addition to color, the distance of clump from the edge of the bog displayed positive correlation with the amount of midge larvae found in pitcher plants (Fig.2). The clump size which used diameter as an indicator did not correlate with the amounts of midge larvae ($p>0.05$) or mosquito larvae ($p>0.05$).
Figure 1. Correlations between mosquito larvae, midge larvae and proportion of pitcher plants with Color 4 in a clump.

There is a positive correlation between the diameter of pitcher plant peristome and the presence of mosquito larvae (Fig 3.), midge larvae (Fig. 4) and prey items (Fig.5). Mosquito larvae and midge larvae tended to be present together (Fig. 6).

Figure 2. The correlation between distance from bog and amount of midge larvae.
Figure 3. Correlation between Mosquito larvae and diameter of pitcher plant.

Figure 4. Correlation between midge larvae and diameter of pitcher plant.
Figure 5. Correlation between dead prey and diameter of pitcher plant.

\[ y = 0.154x - 1.246 \]

\[ R^2 = 0.073 \]

\[ p = 0.017 \]

Figure 6. Number of mosquito larvae versus number of midge larvae.

\[ y = 1.735x + 8.057 \]

\[ R^2 = 0.206 \]

\[ p = 0.000 \]
DISCUSSION

Previous studies have shown a correlation between red-venation and potential prey items (Newell et al. 1998), but we did not find this to be true in our examination of pitcher contents. The positive correlation between color class 4 (mostly red, with some green) and mosquito and midge larvae (Fig. 1), may or may not be working under the same mechanism as prey attraction. Prey may be attracted to a pitchers largely due to abundance of extra-floral nectaries, which are associated with pitcher lips and veins (Plancho, 2007) and other ultra-violet patterns within their sight spectrum (Moran, 1996). Nectar further contributes to their demise by creating a slippery pitcher, which the prey item has difficulty escaping (Bauer et al, 2008).

While the red-venation of pitcher plants has been likened to the appearance of rotting meat and may be initially attractive to an ovipositing dipterans, the attractant for potential inquilines may be more complicated. The decomposition of prey-items within a pitcher could...
serve as an olfactory attractant or deterrent. Mosquitoes were observed to avoid ovipositing in pitchers with highly enriched nutrient levels (Hoekman et al., 2007). This is in support of our observation and speculation that pitchers with too many prey items quickly depleted their oxygen availability and created a toxic environment for inquilines.

The positive correlation of midge and mosquito larvae occurring together (Fig. 6) is further support of their contribution and role in the digestion of prey items within the pitcher. Mosquitoes were not observed to be selecting pitchers with relative midge larvae for oviposition (Heard, 1994) yet their abundance in relation to midge, may be an indication of larval survival. Midge larvae are known to facilitate mosquito growth (Heard 1994). Figure 7 is a testament of the mosquito and midge positive correlation to one another, rather than the common variable of large sized pitchers, for which they both showed preference.

Larger pitchers, based on diameter of peristome were associated with higher midge, mosquito and prey item counts than smaller pitchers (Figs. 3-5). Larger peristome size is generally associated with larger pitchers and an increased ability to hold more water and relative inquiline constituents. Larger pitchers are also less likely to desiccate during dry periods (Kingsolver 1979), and because of their size may be competitive visual targets.

The higher density of midge larvae as distance from the edge of the bog increased (Fig. 2) could be related to life-cycle details of *Metriocnemus knabi*, but it is curious as to why a positive correlation was not also exhibited by mosquito larvae, as the two species appeared correlated in other regards. More information on the life-cycles and oviposting preferences is necessary in deciphering these relationships.

While our results did not indicate a conclusive relationship between coloration and abundance of mosquito and midge larvae, it did hint at a preference for specific variables. Size, distance from the edge of the bog and color class 4 all relay positive correlations with one or more of the concerned items. Further research into the interpretation of these characteristics relative to the observed communities would be advantageous in concluding preferences versus tolerability.

**ACKNOWLEDGMENTS**

Special thanks to Dave Allen for providing technical and logistical support and Ivette Perfecto and John Vandemeer for assisting in the location of Hidden Lake Bog and guiding scientific exploration into this project.


COMPARISON OF TERRESTRIAL SNAIL DIVERSITY IN MIXED FOREST AND MANIPULATED RED PINE (*Pinus rubra*) FOREST

BINBIN LI, RACHEL CABLE, ANA MARCELA CHARA, MINGQI LIU, IMAN SYLVAIN

ABSTRACT

Plant community, calcium availability and moisture are considered to be the main factors restrict the distribution and diversity of terrestrial snails. However, there was lack of studies in temperate land snails. We compared the richness and abundance of terrestrial snails in hickory-oak mixed forest and pure red pine (*Pinus rubra*) forest in E.S. George Reserve. Logs were selected as study area where snails mainly depend on the woody parts and fungi living on decaying logs as food. Our result strongly supports the hypothesis that snail biodiversity is low in pine monoculture which may resulted from low plant diversity and soil acidity. The abundance of snails was also influenced by the decomposition status of logs which is more important than moisture in this area.

INTRODUCTION

Snails are very important members of both terrestrial and aquatic ecosystems. Snails aid in the decomposition of course woody debris and eat fungi, thus contributing to nutrient cycling. Snails serve as prey for many animals. Snails can also be intermediate hosts and vectors of pathogenic parasites which can infect humans. Studying the biology of snails is important for a variety of reasons. For example, some snails, like *Arianta arbustorum* can be used as bio-monitors and indicators of pollution or metal contamination because they can accumulate copper, cadmium, lead and zinc (Burkhard 1993). Despite the relevance of snail biology and natural history, few studies have investigated determinants of terrestrial abundance and diversity in the Great Lakes region. This study sought to determine whether snail abundance and diversity differs in a conifer plantation and a natural deciduous forest.

The main factors thought to influence snail populations are soil moisture, pH, and the availability of calcium. Previous studies have shown that snail abundance and diversity is correlated with soil characteristics such as pH and calcium content in the organic layer. Calcium carbonate uptake is required for snail shell growth, reproduction and withstanding desiccation (Kappes 2006, Machin 1967), and the solubility of calcium carbonate in soil is dependent on pH. At a lower pH level, calcium bicarbonate is converted to calcium carbonate due to cation exchange. Field studies in Southwestern Germany showed that soil moisture was the strongest determinant of snail density and species richness in an undisturbed woodland site, and soil moisture and pH
were closely correlated effects in intermediate or wet soils. In addition to moisture and pH, other habitat characteristics such as plant community gradients and forest types and litter quality are also important factors influencing snail abundance and species dominance (Locasciulli 1987, Martin 2004).

A final consideration of this study was the ways in which humans affect snail populations. Abundance of terrestrial snails may also show responses to deforestation. Theoretical biogeographic distribution studies of gastropods show that land-use practices and anthropogenic disturbance results in nestedness and clustering of gastropod populations. Recently disturbed sites show reductions in species composition (Bloch 2007). We examined whether greater snail diversity and abundance was found in forests which have remained intact on the E.S. George Reserve, in comparison to the manipulated red pine (Pinusrubra) forest. We hypothesized that the greatest abundance and diversity of terrestrial snails would be found in the deciduous forest, due to its neutral pH, greater availability of water, and heterogenous habitat. We predicted that due to chemical interactions between calcium carbonate, low pH due to effect of pine needles on soil chemistry, and homogeneity of plant species, fewer snails would be found in the pine forest.

**METHODS**

This study was conducted on October 9th at the E.S. George Reserve. Two sites were selected: Site 1(Yellow star in Figure 1) is an oak-hickory mixed forest. Site 2(red star in Figure 1) is a pure pine forest. Six logs were randomly select in site 1 and site 2. For each log, we did a 3 meter transect and collected all the snails on the log. Also, we recorded the pH value, soil moisture and decomposition status of each log. We defined decomposition status 1 as slightly decomposed which we could hardly take off the barks, status 2 as medium decomposed which we could easily take off the barks but the inner part of log still remained hard, status 3 as largely decomposed which bark could be easily taken off and the inner part was soft and easy to take apart as well. We identified all the species and whether they were dead or alive following the field guide of snails in Great Lake Region using microscope and recorded the abundance of each species. We used spearman correlation analysis, chi-square and regression for statistics.
RESULT

We collected a total of 154 individuals of four different species in the two sampled forest types. 141 individuals were collected in the mixed forest while 13 were found in the pine stand. The mixed forest also presented higher richness of snail species, as only one species was found in the pine stand (Figure 2, Figure 3).
No significant differences were found regarding the decomposition level of the trunks sampled in the mixed forest and the pine stands (p > 0.01). We found higher abundance in the less decomposed trunks in the two sites (Figure 4), and higher richness of snail species in the less decomposed trunks in the mixed forest (Figure 5).
Figure 5. Snail richness in trunks with different decomposition levels in the mixed forest.

The moisture level was also similar between the two forest types (p>0.01). Abundance of all the snails species tended to decrease with the increase in moisture (Figure 6), but the regression value was not significant (p = 0.74). The same tendency was observed when we examined the two most important species separately (Figures 7-8), but again, the regression was not significant in both cases (*Anguispira alternate*: p = 0.57; *Striatura milius*: p = 0.54).
We also found a negative correlation between the moisture level and the richness of snail species in the mixed forest (Figure 9), and in this case the regression value was marginally significant ($p = 0.07$).

**Figure 6.** Relation between snail abundance and the different moisture levels in the two sampled sites.

**Figure 7.** Relation between abundance of *Anguispira alternate* and the different moisture levels in the mixed forest.
Figure 8. Relation between Abundance of *Striatura milius* and the different moisture levels in the sampled sites.

![Graph showing the relation between abundance and moisture levels. The equation is $y = -4.015x + 16.55$ with $R^2 = 0.038$.]

Figure 9. Relation between snail species richness and moisture levels in the mixed forest.

![Graph showing the relation between richness and moisture levels. The equation is $y = -1.228x + 4.272$ with $R^2 = 0.598$.]

There was a significant positive correlation between the level of the decomposition of the trunks and the level of moisture ($p = 0.03$) (Figure 9). Furthermore, there was a marginal negative correlation between the pH and the level of decomposition of the trunks ($p = 0.07$) (Figure 10).
**Figure 10.** Relation between the level of decomposition of the trunks and the moisture in the two sampled sites.

![Graph showing the relationship between moisture and level of decomposition with the equation \( y = 0.381x + 0.779 \) and \( R^2 = 0.233 \).]

**Figure 11.** Relation between the level of decomposition of the trunks and the pH in the two sampled sites.

![Graph showing the relationship between pH and level of decomposition with the equation \( y = -0.05x + 7.05 \) and \( R^2 = 0.166 \).]

**DISCUSSION**

A much higher species richness and abundance of land snails was found in the oak-hickory forest transects than in the pine transects, indicating that some environmental factor is indeed limiting
the snails in the pine forest but not in the oak-hickory forest. One possible explanation is the lack of available calcium carbonate (lime) in the acidic soil conditions where pine forests are found. Lime is required for snails to build their shells (Fournie and Chetail 1984), but it acts as a buffer in these acidic conditions (McLaughlin and Wimmer 1999) and can dissociate to its cationic form that cannot be used by the snails and leaches out of the soil very easily. These low pH values and low available calcium values have been shown to yield low land snail species richness and abundance (Hotopp 2002, Martin and Sommer 2002).

Such lowered amounts of lime may be tolerated by certain species of snails with lower calcium requirements. Striatura milium seems to be one of these species, as it was the only species found in the pine forest, and we noticed it had a thinner shell than the other species found in the oak-hickory forest. This is further evidenced by the fact that of the two low pH logs we surveyed, one in the oak-hickory forest and one in the pine stand, both of these logs were only inhabited by S. milium. Striatura milium is known as an abundant species through several habitats in the E.S. George Reserve and the Great Lakes Region with soil pH values as low as 5.0, indicating that it is indeed a tolerant species (Archer 1939, Nekola 2003 (1), Nekola 2003 (2)).

We also noted that the transects with lower decomposition ratings had higher abundance and species richness. This may be because more highly decomposed logs will have less nutrients available to the snails and any microbes they might be eating, as reported by Hotopp (2002) with low land snail abundances in highly decomposed leaf litter. A similar trend occurred in comparing the moisture of the transects to the species abundance and richness, where more decomposed logs tended to be more moist, and so we found moisture to be an indicator of decomposition rather than a preferred factor for land snail habitation. High moisture also corresponded to low pH values in the two transects with pH 6.7, which we have already mentioned were only occupied by low abundances of S. milium.

Although we did see a difference in species richness and abundance between the oak-hickory forest and pine forest, we did not see a significant difference in pH values of transects between the two areas, as the two lowest pH logs were from both areas. One reason for this may be that all pH and moisture measurements were taken on the log being surveyed, while a better indicator of the differences between the two areas would have been the soil pH and moisture surrounding the log. The difference in species diversity and abundance between the forests might be a result of the soil pH or moisture, as land snails lay their eggs in the soil (Cowie 1984) and only feed on the logs once they have hatched. Direct measurements of available calcium levels would also improve upon our survey and data, instead of assuming a correlation between pH values and available calcium.
ACKNOWLEDGEMENT

We would like to thank Thomas Duda for his instruction and suggestion in this project.

REFERENCES


THE EFFECT OF NUTRIENT ADDITION ON SPECIES ABUNDANCE AND DIVERSITY IN TWO EXPERIMENTAL PONDS AT THE E.S. GEORGE RESERVE

IMAN SYLVAIN, CINDY BICK, SAHAR HAGHIGHAT, AND WILLIAM WEBB

ABSTRACT

It is well known that nutrient addition in aquatic systems can result in greater primary productivity. What is less understood is how the addition of nutrients in aquatic systems determines the community structure of plants and animals within those systems. A question of particular interest is whether aquatic ecosystems with added nutrients have greater biodiversity. This study investigates precisely that inquiry. Differences in community structure were examined in two experimental ponds at the E.S. George Reserve. Plant and animal species diversity and abundance were compared in a pond known to have received nutrient enrichment through the decomposition of a deceased muskrat, and a pond without known nutrient addition. Plant and animal specimens were collected and/or identified along three 6.5 meter transects which ran from the center of the pond to the vegetative edge. Results of this study show that the plant and animal communities within the ponds were relatively similar despite differential nutrient addition. Nutrient enrichment did not have a significant effect on the levels of biodiversity within each pond. These findings suggest that nutrient addition alone is not a determinant of high biodiversity in aquatic ecosystems.

INTRODUCTION

Aquatic ecosystems have important ecological functions. Examples of these functions include nutrient recycling, water purification, flood attenuation, recharging of ground water supplies, and the provision of habitats for numerous organisms (Karickhoff 1984, Brix & Schierup 1989, Junk et al. 1989, Alberti & Marzluff 2004). Aquatic ecosystems are often categorized as either marine or freshwater ecosystems (Starr et al. 2009). This study was conducted in two freshwater experimental ponds at the E. S. George Reserve. Previous studies have described the effect of nutrient enrichment on species abundance and diversity in aquatic ecosystems (Lane & Levins 1977, Cederholm et al. 1999, Hocking et al. 2009). Variations in nutrient levels are integral to our understanding of the ecological mechanisms that lead to the stability and functionality of aquatic ecosystems. A shortage of nutrients can limit vegetative and macroinvertebrate growth in aquatic systems, while excess nutrients can lead to eutrophication (Vitousek & Howarth 1991, Smith et al. 1999).

Nitrogen and phosphorous are two critical components of aquatic ecosystems. These nutrients control the rates of photosynthetic processes which can determine the total growth of the system (Smith 1984, Vitousek & Howarth 1991). The growth of aquatic plants and algae is often limited by short supplies of nitrogen and phosphorous (Vitousek & Howarth 1991). The
primary natural sources of nitrogen and phosphorous in ecosystems are the decomposition of organic matter and the gradual dissolving of minerals in soils (Rabalais 2002). Examples of anthropogenic sources of nitrogen and phosphorous are sewage, fertilizers, and animal wastes, which flow into aquatic systems (Rabalais 2002).

Studies by Cederholm et al. (1999) and Hocking et al. (2009) suggests that the addition of organic materials, such as salmonid carcasses, contribute essential nutrients and energy for the development of aquatic and terrestrial ecosystems. Our research was prompted by a similar process that occurred in one of the experimental ponds at the E. S. George Reserve. Additional studies (Yee & Juliano 2006) have shown that aquatic systems receiving invertebrate carcasses exhibit higher levels of nitrogen and phosphorous input compared to systems with plant-derived detritus. Detritus from animal sources can affect food webs because it is readily available and ingested by various consumers in aquatic systems. Plant-derived detritus, by comparison, is indirectly available to top consumers because trophic level decomposers are needed to break down essential nutrients (Yee & Juliano 2006). Further support for this is given by Wipfli et al. (1998), who reported an increase in species abundance and diversity of macroinvertebrates in carcass-enriched streams in Alaska. Although our study does not focus on the levels of nitrogen or phosphorous in the ponds, we hypothesized that the addition or elevated levels of nitrogen and phosphorous from animal-derived detritus increased the plant production. We also predicted that the increase in primary production in the nutrient enriched pond consequently gave rise to greater animal species abundance and diversity.

METHODS AND MATERIALS

Study Site

This study was conducted in two experimental ponds at the E.S. George Reserve. The George Reserve is located in Livingston County, Michigan, roughly 25km from Ann Arbor (42° 28' N, -84° 00' W). The reserve contains 525-hectares of private land. In the northeast corner of the reserve there are 12 small, man-made experimental ponds. The ponds were constructed in 1987 to conduct long-term amphibian studies. The ponds are 15 meters in diameter and 1 meter deep. The ponds were colonized by plants as a result of natural primary succession, and were not manipulated in terms of vegetation. A muskrat was known to have died in one of the twelve ponds in 1988, making that particular pond very distinctive.
Vegetation and Animal Surveys

Comparisons of the plant and animal communities were made by sampling the pond with added nutrients to a more standard pond. The standard pond was chosen using a random numbers generator. To analyze plant species abundance and diversity in the ponds, three 6.5m transects were set up from the origin of the pond leading to the vegetative edge. Every plant found along the transect was recorded. The plant’s length along the transect line was measured to estimate the percent cover and relative importance of each species. Plants were identified to species or morpho-species level using various plant keys, including Michigan Flora by Ed Voss. Aquatic insects, mollusks, and amphibians were sampled along the three 6.5m transect lines using dip nets and sift pans. 10 dips were sifted through per transect collection. Collected animals were identified to genus, or species level if possible.

Water Chemistry

Water temperature and pH were measured in the center of each pond using a standard thermometer and pH meter.

RESULTS

Table 1. List of plant species found in each pond and associated percent coverage.

<table>
<thead>
<tr>
<th>Pond</th>
<th>Species</th>
<th>Percentages/Importance</th>
</tr>
</thead>
<tbody>
<tr>
<td>&quot;Experimental&quot; Pond 4</td>
<td>Phragmites australis</td>
<td>100.0</td>
</tr>
<tr>
<td></td>
<td>Lemna minor</td>
<td>46.1</td>
</tr>
<tr>
<td></td>
<td>Carex comosa</td>
<td>17.5</td>
</tr>
<tr>
<td></td>
<td>Vitis ripera</td>
<td>17.0</td>
</tr>
<tr>
<td></td>
<td>Bidens cernus</td>
<td>27.5</td>
</tr>
<tr>
<td></td>
<td>Epilobium palustre</td>
<td>18.7</td>
</tr>
<tr>
<td></td>
<td>Lycopus uniflorus</td>
<td>15.2</td>
</tr>
<tr>
<td></td>
<td>(vegetative basal rosette)</td>
<td>1.8</td>
</tr>
<tr>
<td></td>
<td>Typha argustifolia</td>
<td>0.3</td>
</tr>
<tr>
<td></td>
<td>(vegetative Brassicaceae)</td>
<td>0.1</td>
</tr>
<tr>
<td>Pond 7</td>
<td>(vegetative aquatic plant)</td>
<td>100.0</td>
</tr>
<tr>
<td></td>
<td>Typha argustifolia</td>
<td>100.0</td>
</tr>
<tr>
<td></td>
<td>Salix amygdalides</td>
<td>33.5</td>
</tr>
<tr>
<td></td>
<td>Typha glauca</td>
<td>27.4</td>
</tr>
<tr>
<td></td>
<td>Lemna minor</td>
<td>19.9</td>
</tr>
<tr>
<td></td>
<td>(vegetative Graminae)</td>
<td>7.7</td>
</tr>
</tbody>
</table>

Table 1 shows the comprehensive list of the plant species found in both ponds. The table also shows the percent cover of each species within the pond. This calculation was made by setting the total length of each plant along the transect as the radius of a small circle. In this manner we were able to determine the relative area of a specific plant within the pond. The relative area coverage of each species was set as a ratio of the total pond area to determine the species’ percent cover. The most dominant plant species in the Experimental Pond 4 was Phragmites australis (invasive genotype), which was found throughout the pond (100% coverage). Lemna minor was also highly abundant in Pond 4, covering 46% of the pond. The Standard Pond 7 was
dominated by *Typha angustifolia* and an unidentified vegetative aquatic plant (both 100% coverage).

**Figure 1.** The total number of animals found within each pond.

![Number of Animals within Each Pond](image)

Figure 2 shows the total number of individual animals found within each pond. This graph represents the total abundance of aquatic insects, mollusks, and amphibians. It appears that the Standard Pond 7 has a greater number of animals living in it than the Experimental Pond 4.

**Table 2.** List of animal species found in each pond.

<table>
<thead>
<tr>
<th>'Experimental' Pond 4</th>
<th>Pond 7</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Rana clamitans</em> (Green frog)</td>
<td><em>Libellulidae</em> (dragonfly)</td>
</tr>
<tr>
<td><em>Rana catesbeiana</em> (Bull frog)</td>
<td><em>Aeshnidae</em> (dragonfly)</td>
</tr>
<tr>
<td><em>Hirudinae</em> (leech)</td>
<td><em>Bivalvia</em> (mollusk)</td>
</tr>
<tr>
<td><em>Physidae</em> (snail)</td>
<td><em>Planorbidae</em> (snail)</td>
</tr>
<tr>
<td><em>Lymnacidae</em> (snail)</td>
<td><em>Hirudinae</em> (leech)</td>
</tr>
<tr>
<td><em>Dytiscus</em> (beetle)</td>
<td></td>
</tr>
</tbody>
</table>

Table 2 shows the animal species that were found in each pond. Experimental Pond 4 had one more species collected than Pond 7. The Standard Pond 7 was home to aquatic insects, mollusks, and leeches. In addition to these animals, the Experimental Pond 4 was also home to two amphibian species as well.
Figure 2. Plant diversity species accumulation curves in both ponds.

Figure 3. Animal diversity species accumulation curves in both ponds.
Figure 3 and 4 show species accumulation curves of plant and animal diversity within both ponds. Overlapping confidence intervals on the graph show that the two sites were not significantly different from each other (p > 0.05). There was no statistical difference between the extrapolated species richness of either plant or animal diversity in the ponds.

Water chemistry tests showed that the pH in both ponds was similar. The pH in the Experimental Pond 4 was 6.75, and 6.5 in the Standard Pond 7. The temperature in the ponds differed by only ½ a degree, with results of 10.5°C in Experimental Pond 4, and 11°C in the Standard Pond 7.

DISCUSSION

Results of this study show that there were not stark differences in the plant species compositions of the two experimental ponds, despite the addition of nutrients. The pond which experienced nutrient addition appeared to support more plant life based upon greater biomass in the pond, but this did not translate into greater diversity of plant species. Although the level of plant diversity in Experimental Pond 4 was not significantly different from the Standard Pond 7 in total number of species, the experimental pond did support unexpected plant life. The plants present in the nutrient enhanced pond included terrestrial (that is, non-aquatic) flora, such as *Epilobium* sp. and *Vitus* sp., which were not present in the standard pond. In addition to these terrestrial plants, aquatic flora such as *Lemna minor*, *Phragmites australis*, and *Typha angustifolia* were also present in the Experimental Pond 7. Studies have shown that nitrogen is a limiting growth factor for plants, even in aquatic systems (Penderson 1995), and the addition of nitrogen to aquatic systems can result in greater primary productivity (Lane 1977). Interestingly, we did not find an associated higher abundance of plants in the nutrient enriched pond. We also did not observe the expected trend for animals in the pond with added nutrients. In fact, we discovered exactly the opposite.

The prediction that greater animal abundance and diversity would be found in the pond with added nutrients was incorrect. The prediction assumed that animals would prefer the Experimental Pond 4 to Standard Pond 7 due to the greater availability of food and habitat in a more heterogeneous vegetative landscape, but this was not observed. Results similar to ours have been shown by Dodson et al. (2000) who reported the greatest animal biodiversity in lakes with low primary productivity. Despite the lower number of animals found within the Experimental Pond 4, the presence of amphibians compounds our data analysis. Anurans are known to have strong effects on the partitioning of the flow of nutrients in aquatic systems through their interactions with phytoplankton and periphyton, which then effects zooplankton and insects (Wilbur 1997). It is possible that amphibians show a greater preference for ponds with higher nutrient contents for trophic reasons, as opposed to habitat quality. Thus the cycle involving amphibians and phytoplankton can continue to add nutrients to aquatic systems, which plants utilize by indirect benefit.

Of course, another possible explanation for our animal data could be related to seasonality and the uptake of finite resources in aquatic systems. Previous studies have shown that some benthic dwellers, in particular mollusks, can experience population flux during times of varying nutrient levels in the water column (Maar et al. 2010). Of course, attempts to qualitatively or quantitatively analyze current community structure to infer productivity and nutrient enhancement in aquatic systems can be flawed. Current species abundances may reflect both the differential growth of individuals within the ponds, as well as species compositions over
the last 20 years. These factors are hard to distinguish in short-term surveys. Our data indicates that despite the differences in biomass within each pond, overall species richness did not vary significantly between the ponds of different nutrient levels. This study shows the complexity of species composition, abundance, and the construction of biodiversity in nutrient-limited aquatic systems.

ACKNOWLEDGEMENTS

We would like to our professors, John Vandermeer and Ivette Perfecto for giving us the freedom to explore exciting questions for our group projects. We would also like to thank Dave Allen for insight into the methodology used in this study and providing us with needed supplies.

WORKS CITED


DIFFERENTIAL RESPONSE TO MICROHABITAT VARIABILITY IN SEEDLING GROWTH STRATEGIES OF EARLY AND LATE SUCCESSIONAL SPECIES

WILLIAM WEBB, DEBRA GOLDBERG, DAVID GONTHIER, JOHN GUITTAR, KASSANDRA SEMRAU, KATE ZEMENICK

Abstract. Microhabitat conditions and resource availability will influence the growth and survivorship of all individuals within a population. Seedling growth strategies will presumably have variability in the response to the different abiotic and biotic factors surrounding them. Our investigation focused on four sub-canopy species (*Acer rubrum*, *Prunus serotina*, *Hamamelis virginiana*, and *Carya spp.*) of varying shade tolerances and successional strategies and how patterns of growth related quantitatively with different levels of canopy cover, leaf litter, and microhabitat composition. We observed positive and inverse correlations (P < 0.05) of mean seedling height in Hickory and Red Maple species respectively to denser canopy cover. Total number of individuals of all species of seedling decreased with increasing levels of shrub cover (P=0.001). Preference for lower biomass of leaf litter was also uncovered in Black Cherry (P < 0.05) and Red Maple seedlings (P < 0.01).

INTRODUCTION

Numerous factors can affect forest regeneration, including abiotic factors such as soil characteristics and light availability (Grubb, 1977) along with disturbance, yearly reproductive success, seed dispersal, etc. Many hypotheses have sprung up to explain discrepancies in the variable distributions of adult trees and seedlings within the same landscape. One such explanation, the gap partitioning hypothesis, predicts that gap openings favor seedling recruitment indiscriminately for all species (Denslow, 1980). However, each species’ differential response to their environment can result in variation in distributions across the landscape (Denslow, 1980). The local environmental conditions can change drastically in the amount of time it takes for a seedling to become a successful reproductive adult through disturbances. These changes in the local environment can alter how conducive the conditions are for a particular species during its establishment stage.

Early successional species take advantage of disturbance in their habitat and are often shade intolerant (Walters & Reich, 1996). Late successional species are more often shade tolerant and able to persist on the forest floor until growing conditions improve (Spurr & Barnes, 1980). There is a trade-off between low-light survival and high-light growth rate which tends to balance the proportion of early and late successional species (Kobe et al, 1995, Pacala et al, 1996). Shade tolerant species must maximize their use of soil nutrients to compensate for the lack of sunlight, and often wait for gaps in the canopy before investing large amounts of resources in growth. Additionally, plant growth is limited by nitrogen availability. Decomposing leaf litter deposits nitrogen and other compounds into the soil, providing much-needed nutrients for plants and other organisms. It follows that the abundance of leaf litter in forests might be reflected in the
distribution of seedlings, and that areas with high quantity of leaf litter would contain
greater numbers of seedlings. The size of seeds may also have an effect on the
distribution of seedlings in the local environment. Large seeds have greater success
emerging from deep leaf litter, therefore we expect species with large seeds to be more
frequent than small-seeded species in areas of high leaf litter. Areas with low levels of
leaf litter are more likely to dry out, and we expect a positive correlation between leaf
litter and the number of seedlings. In this study we examine the relationship between
seedling germination and growth with canopy cover and leaf litter abundance. We
categorize red maple (Acer rubrum) as a small-seeded shade tolerant species, hickory
(Carya spp.) as a large-seeded shade intolerant species, black cherry (Prunus serotina) as
a small-seeded shade intolerant species, and witch-hazel (Hamamelis virginiana) as a
small-seeded species (Martin & Gower, 1996).

MATERIALS AND METHODS

The seedling survey was carried out on October 9th, 2010 at the Edwin S. George
Reserve near Pickney, MI. Two areas were selected for the census, both in the Big
Woods Plot, a 16 ha long term forest dynamics study site. The first area was located in a
kettle, and the second along an esker. Therefore, species composition and light levels
differed between the two areas.

At each of the two sites, 0.25m² quadrat was randomly placed on the ground 50
times. The first site, being sparse of seedlings required a less random placing, where we
would first locate a patch of seedlings, then haphazardly place the quadrat. The second
site was more plentiful of seedlings, and the quadrat was tossed randomly to obtain the
sample.

Each time the quadrat was placed, the canopy cover was recorded at each corner
with a densiometer. The 4 measurements were then averaged to obtain an estimate of
canopy cover for the entire square. As the densiometer data was collected while standing
erect, it was necessary to estimate the understory cover of the quadrat. This was done by
standing above the quadrat to obtain a bird’s eye view, and then estimating the
percentage cover due to shrubs. Whenever possible, the species of shrub was noted. The
leaf litter abundance was also estimated within each quadrat. This was done by
haphazardly tossing a Stonybook Farms Yogurt lid, and estimating the percentage leaf
cover within the circle. Each seedling within the quadrat was identified, and the leaf litter
around it was estimated in the same way; the Yogurt lid was placed around the seedling
so that the seedling stem would be at the center of the circle, and then the leaf litter cover
was estimated.

Statistical analysis was done using Microsoft Excel and R.

RESULTS

Plot level comparisons
Seedling species differed in their relationships with canopy cover. Hickory seedling height increased with canopy cover (Fig. 1, $R^2 = 0.2$, $P = 0.03$). Red maple seedling height was inversely related to canopy cover (Fig. 2, $R^2 = 0.1$, $P = 0.018$). Black cherry and witch hazel seedling height was not different across canopy covers.

Figure 1. The relationship between hickory height and canopy cover (%).

Figure 2. The relationship between red maple height and canopy cover (%).

Shrub cover also had an important influence on seedling height. Hickory, black cherry, red maple, and witch hazel seedling heights were un-related to shrub cover. However, the total number of seedlings per plot decreased inversely with shrub cover (Fig. 3, $R^2 = 0.1$, $P = 0.001$). However, each species individually had no significant relationship.

Figure 3. The relationship between total number of seedlings per plot and shrub cover (%).
Leaf litter cover had differing effects on seedling species establishment. There was 11% less litter cover at the base of black cherry seedlings than at random points within plots (Fig. 4, t = 2.1, df = 20, p-value = 0.048). Red maple followed a similar trend, there was 12% less litter cover at the base of red maple seedlings than at random points within plots (Fig. 5, t = 2.8, df = 55, p-value = 0.007). There was no difference between the leaf litter cover at the base of hickory and witch hazel seedlings and random.

There was no difference of the living plant cover at the base of seedlings and at random points within plots.

Figure 4. Percent leaf litter cover at the base of black cherry seedling and at random points within plots.

Figure 5. Percent leaf litter cover at the base of red maples seedlings and at random points within plots.
DISCUSSION

Our study indicates a correlation between microhabitat and seedling establishment and growth. The concurrent increase in Hickory height and decrease of Red Maple height under greater canopy cover support the idea that heterogeneity - in our case differences in fall light levels - may facilitate the successful establishment of different species. Potentially, Hickory seedling growth may be favored in patches with less light and more coverage. This finding supports experimental observations that chlorophyll and seedling height were negatively related to available light (Robison & McCarthy 1999). While research has shown that Red Maple seedlings can germinate in a wide shade gradient (Abrams 1998), our data support the growth, but not establishment, of Red Maples in relatively higher light. This supports older data characterizing Red Maples as only moderately shade-tolerant (Burns and Honkala 1990). It is important to note that our study reports only a correlation of species to measured light levels. Many alternative mechanisms may dictate this correlation, such as secondary dispersal, herbivory, or seed predation. Lastly, light filtration to the forest floor varies seasonally. Conditions in October do not indicate conditions during germination, which thus may lead to incorrect conclusions.

In support of previous findings at the E.S. George Reserve (Hyduke et. al. 2010 Field Course), we observed a ubiquitous decrease in seedling number with increasing canopy and/or shrub cover. As previously hypothesized, this could be due to resource light availability; however in the case of Hickories a lack of resources is not logical as Hickory seedlings were taller. An alternative explanation could include common shelter-seeking small mammals like Peromyscus leucopus. Small mammals may be consuming more Hickory seeds in sheltered areas, but the surviving Hickories thrive because Hickory is a shade-prefering species. More research is needed to investigate this potential indirect effect of shading for Hickories.

Black Cherry and Red Maple showed preference for microhabitats with less litter. This could indicate one or both of the following: 1) seedlings preferentially germinated in litter gaps, or 2) litter did not settle as often at the base of seedlings. While the former is
more interesting it is not rigorously testable. Since litter was measured in mid-October, spring litter present during seedling germination was likely already in the process of decomposition. The latter explanation – that falling leaves did not settle as densely around seedlings – is a more likely to be responsible. A field example teasing apart spring and fall litter is necessary to further understand its role in seedling development.

ACKNOWLEDGEMENTS

We would like to thank David Allen for his logistical insight and guidance to a site most suitable for conducting this study.

LITERATURE CITED


Grubb, P. 1977


Abstract: Although much is known about the biochemistry and physiology of deciduous color change in temperate trees, its adaptive value remains poorly understood. A variety of environmental gradients have been suspected to influence leaf color within single species, including but not limited to temperature, sunlight, soil moisture, and soil quality. We conducted the present study to assess the differences in the leaf color patterns of some common tree species in the George Reserve, and relate them to the conditions of light intensity and canopy cover in the forest. Furthermore, we assessed the ratio of colors of single leaves to test their use as a possible indicator of the ratio of colors in the whole tree.

INTRODUCTION

Cool temperate forests are characterized by deciduous plant species which undergo a seasonal fall change in leaf color that is followed by leaf shed. During this process the increase of the night length causes the chlorophyll production to be slowed and eventually stopped. As chlorophyll content falls, other pigments present in the leaves such as xanthophylls, carotenoids, and anthocyanins are revealed (Schaberg et al. 2008, Koike 1990).

Although much is known about the biochemistry and physiology of this color change, its adaptive value remains a mystery. Although xanthophylls and carotenoids are produced throughout the growing season, anthocyanin pigments are often synthesized toward the end of a leaf’s lifespan and require a metabolic cost (Schaberg et al. 2008). Two main theories have been proposed to explain the process: 1) the photoprotection hypothesis, which argues that red pigments protect against the harmful effects of light at low temperatures and 2) the coevolution hypothesis, which argues that red is a warning signal toward aphids attempting to feed on these leaves, that they are well defended by chemical deterrents and therefore not good food. Then the aphids’ offspring would not lay their over-wintering eggs on the trees which would have the selective advantage for the trees of reducing the damage done to their new growth by aphids hatching from those eggs in the spring (Archetii et al. 2008). A third possibility is the null hypothesis, that the color of some autumn leaves is just a coincidence as trees prepare for the winter, and the colors are only a molecular by-product of this process.

The existence of a relationship between the pattern of autumn coloring and the growing strategy of a tree species has been discussed by several authors. Early-successional species have a longer period of leaf emergence than late-successional species and mid-successional species show intermediate characteristics (Koike 1990). Temperature, sunlight, and soil moisture are also potential factors influencing the variation of leaf color change during the fall season even within species. Abundant sunlight and low temperatures after the time the abscission layer forms cause the
chlorophyll to be destroyed more rapidly. Cool temperatures, particularly at night, combined with abundant sunlight, promote the formation of more anthocyanins. Freezing conditions interrupt the formation of anthocyanins and drought stress during the growing season can sometimes trigger the early formation of the abscission layer, so leaves may drop before they have a chance to develop fall coloration (Aker et al. 2009).

A previous study in E. S. George reserve tested the photoprotection and the coevolution hypothesis of some tree species, and found evidence to support the first for the red maple, but could not find significant evidence to support or reject the second (Gan et al. 2008). A posterior study assessed the relation between leaf color patterns of trees and their fitness, finding that the timing of coloration has an impact in the growth rate of trees (Gan 2008).

We conducted the present study in order to assess the differences in leaf color patterns of a broader group of species in the E. S. George Reserve, and related them to the conditions of light intensity and canopy cover in the forest. We also examined the distribution ranges of the tree species to know if the different times of abscission were related to the stress of species located on the limit of their distribution range. Furthermore, we assessed the ratio of colors of single leaves as a possible indicator of the ratio of colors in the whole tree.

METHODS

Study site

Our study site is located in the Big Woods of the E.S. George Reserve. The forest canopy is largely dominated by Carya (Hickories), Quercus (Oaks), and to a growing extent Acer rubrum (Red Maple) and Prunus serotina (Black Cherry).

Sampling

To sample a representative area in Big Woods, we qualitatively selected two hectares with different canopy densities and different canopy compositions. One transect was laid 100m north and 100m south of the (-100,100) point in the Big Woods Plot. Surveying 6m on either side of the transect, we qualitatively recorded the percentage of green, red, yellow, and brown leaf coverage of each tree. Percentages were the mean of two independent observers, and observer teams were switched throughout the survey. For the color of the leaves, we recorded the percentage of green (%), red(%), yellow(%) and brown(%) of entire tree. We also classified the canopy coverage of each tree into four categories: 0%-24.9%, 25%-49.9%, 50%-74.9% and 75%-100% canopy covered. We also recorded whether the tree had more than 50% of its leaves remaining or not.

To collect leaves for our quantitative test of color ratios, whenever possible we removed the bottom leaf of each tree within the transect. We photographed each individual leaf using a Nikon D-5000 digital camera, a white paper background, and a small portable shelter to standardize light and distance of photograph. Images were then imported into ImageJ and the color ratios were calculated using the following color thresholds: 0-25 Red/Brown, 25-45 Yellow, 45-100 Green. Each color category was
converted to binary and its area was measured relative to the other colors’ areas, resulting in accurate color ratios.

RESULTS

Of the 1200m² of our transect area, we sampled 123 trees. Dominating our survey was *P. serotina*, with a frequency of almost 50% (Table 1). Trees were found to be in each canopy cover (CC) category, and at varying levels of abscission. Leaf color ranged from entirely green to entirely brown. We collected 43 leaves to be analyzed for pixel ratios.

*Leaf Ratios and Canopy Cover Level*

For all trees surveyed, there was no difference in the mean ratio of green, yellow, brown or red between different canopy cover levels (Figure 1). However, it is possible that individual species that have different phenology of leaf senescence and leaf abscission that are not apparent from the analysis with all trees together. Thus, we looked for differences in the color ratios between CC levels in *Prunus serotina* and *Carya spp.* - the two most prominent species in our study, and the two most common species in both the understory and canopy.

![Figure 1: Leaf Color Ratios in Different Canopy Cover Levels](image)

The ratio of green leaves in *P. serotina* in 0-24% CC was significantly lower than the ratio of green leaves in greater than 25% CC (Figure 2). The ratio of yellow leaves in *P. serotina* in 0-24% CC was significantly higher than in CC >25%. The ratio of yellow was significantly greater than the ratio of green in 0-24%, 25-49% and 50-74% CC. The level of significance decreases with increasing canopy cover, and at 75-100% CC, the significance is lost. This suggests that the yellow to green leaf color ratio decreases with increasing CC; *Prunus serotina* senesces earlier if it is in the sunlight.

<table>
<thead>
<tr>
<th>Species</th>
<th>N</th>
<th>Frequency</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Prunus serotina</em></td>
<td>61</td>
<td>0.496</td>
</tr>
<tr>
<td><em>Carya spp.</em></td>
<td>18</td>
<td>0.146</td>
</tr>
<tr>
<td><em>Amelanchier arborea</em></td>
<td>15</td>
<td>0.122</td>
</tr>
<tr>
<td><em>Quercus velutina</em></td>
<td>13</td>
<td>0.106</td>
</tr>
<tr>
<td><em>Quercus alba</em></td>
<td>8</td>
<td>0.065</td>
</tr>
<tr>
<td><em>Acer rubrum</em></td>
<td>3</td>
<td>0.024</td>
</tr>
<tr>
<td><em>Corylus virginiana</em></td>
<td>2</td>
<td>0.016</td>
</tr>
<tr>
<td><em>Ostrya virginiana</em></td>
<td>1</td>
<td>0.008</td>
</tr>
<tr>
<td><em>Sassafras albidum</em></td>
<td>1</td>
<td>0.008</td>
</tr>
<tr>
<td><em>Ulmus americana</em></td>
<td>1</td>
<td>0.008</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>123</td>
<td>1.000</td>
</tr>
</tbody>
</table>
Unlike *P. serotina*, only one *Carya* surveyed had green leaves. Therefore, it was more useful to look at the ratios of yellow and brown. The ratio of yellow leaves in *Carya spp.* in 0-24% CC was significantly greater than the ratio of brown leaves in greater than 25% CC (Figure 3). The ratio of brown leaves was not significantly different between CC levels, but it strongly tended to increase with increasing CC. The ratio of brown was significantly greater than the ratio of yellow in 0-24%. There was no difference between yellow and brown in 25-49% or 50-74% CC. However, the ratio of brown was significantly higher than the ratio of yellow in 75-100% CC. Thus presenting the trend in *Carya* leaves: a higher ratio yellow in the canopy, and a higher ratio of brown leaves in the sub-canopy. Unlike *Prunus serotina*, *Carya spp.* seems to senesce later when exposed to more sunlight.
Leaf Color Ratios and Amount of Leaf Abscission

In all trees, there was no difference in the ratio of green, yellow, brown or red in trees with more than 50% of their leaves or less than 50% of their leaves. This pattern was consistent with Carya spp. and also with P. serotina. Of the individuals with >50% of their leaves, P. serotina had a 28% higher ratio of green leaves than Carya spp. (p < 0.001). For all individuals, regardless of the amount of abscission, Carya spp. had a 57% higher ratio of brown leaves than P. serotina (p < 0.0001).

Single Leaf Color Ratios and Whole Tree Color Ratios

To see if the color ratios on the single bottom-most leaf would be representative of the color ratios on the entire tree, we analyzed the ratios of green, yellow and red/brown for 4 species: Prunus serotina, Carya spp., Amelanchier arborea, and Quercus alba. In no species did all three colors on the leaf predict the ratio of the three colors on the entire tree. However, each species had at least one color that the leaf predicted reasonably well for the entire tree (Table 2).

Table 2: Linear Regression Results - Whole Tree Leaf Color Ratios vs.Single Leaf Color Ratios

<table>
<thead>
<tr>
<th></th>
<th>P. serotina (N = 19)</th>
<th>Carya spp. (N = 10)</th>
<th>A. arborea (N=6)</th>
<th>Q. alba (N=4)</th>
</tr>
</thead>
<tbody>
<tr>
<td>% Green</td>
<td>R² 0.19 p 0.0619</td>
<td>R² 0.99 p 0.0001</td>
<td>R² 0.82 p 0.0133</td>
<td>R² 0.82 p 0.0944</td>
</tr>
<tr>
<td>% Yellow</td>
<td>R² 0.06 p 0.2952</td>
<td>R² 0.02 p 0.7161</td>
<td>R² 0.01 p 0.8828</td>
<td>R² 0.30 p 0.4536</td>
</tr>
<tr>
<td>% Brown/Red</td>
<td>R² 0.57 p 0.0001</td>
<td>R² 0.01 p 0.7864</td>
<td>R² 0.30 p 0.2589</td>
<td>R² 0.17 p 0.5898</td>
</tr>
</tbody>
</table>

DISCUSSION

When considering all trees together, there is no clear pattern in the ratio of leaf colors across CC levels. However, the two most common species, Carya sp. and Prunus serotina could be confounding this relationship. Since yellow ratio decreased with increasing CC in P. serotina but increased in Carya, these trends could be negating each other. This inverse relationship could indicate different two different tree strategies: P. serotina utilizes high canopy cover to photosynthesize later in the season, and Carya use leaves in low canopy cover to photosynthesize later in the season. However, a number of factors can undermine any conclusions. First, entirely dead leaves hanging on a tree may simply be a product of chance, wind, or stem endurance, and they should not be included in an analysis. Second, as suggested by this study, leaf fall is variable among individuals and species. Thus, our measure of canopy cover is potentially under rapid changes due to leaf fall rates in the canopy trees. Third, many other variables were not investigated in this study, including tree height, girth at breast height, soil quality, moisture, and density.

Nonetheless, our findings lead us to question if the amount of abscission had an effect on leaf color ratios. Leaf color ratios do not differ significantly through leaf abscission, suggesting that senescence is a staggered process for tree species. We found that the P. serotina with early in abscission had a higher ratio of green leaves than Carya spp. Further, for all individuals of the two dominant species, Carya spp. had a 57% higher ratio of brown leaves than P. serotina. This supports our previous hypothesis that P. serotina leaves abscise later in the season than Carya spp., but Carya spp. leaves
asbcise long after senescence is complete. This may be due to the ranges of these two species. The range of *Carya* ends in mid-Michigan, while *P. serotina* is reported to span the entire lower peninsula (Figure 4).

Figure 4: Ranges of *Carya ovate* (left) and *Prunus serotina* (Burns and Honkala 1990)

As for the ability of a single leaf to predict the color ratios on an entire tree, we found mixed results. No single leaf was able to predict the color ratios for a tree in any of the species analyzed. It seemed that different species had different colors that the leaf could predict, and only reasonably well at that. Our study was limited by sample size. For example, *Quercus alba* had only 4 replicates for this analysis. It is likely that with a larger sample size, we could determine if a single leaf could predict tree leaf color ratios. It is also possible that a leaf’s ability to predict tree leaf color ratios is dynamic during leaf senescence. This was not tested in our study, but would be interesting to follow up on. Predicting power would likely increase if more than one leaf was used from each tree, possibly leaves from different exposure to sunlight.

WORKS CITED

Aker S., Bentz S., Clark E., Elias T. The Science of Color in Autumn Leaves. URL = http://www.usna.usda.gov/PhotoGallery/FallFoliage/ScienceFallColor.html


AN ANALYSIS OF BIG WOODS TREE DEMOGRAPHY USING EMPIRICAL DATA AND A SIZE-STRUCTED POPULATION MODEL

RAFAEL D’ANDREA, SUSAN CHENG, JOHN GUITTAR, ABIGAIL HYDUKE, MINGQI LIU, AND YAEL ARTZY-RANDRUP

Abstract. Understanding how growth, mortality, and recruitment affect tree demography provides insight into how forest structure has changed and may change in the future. The purpose of this project was to calculate the distribution of sizes for dominant tree species in the Big Woods and to determine whether a model could predict those patterns. Results show that the model can predict the size distributions for *Prunus serotina*, *Acer rubrum*, and *Hammamelis virginiana* but not the entire distribution for *Quercus alba*, *Quercus rubrum/veluntina*, and *Carya spp*. These results imply that tree species with predictable distribution curves may be in a state of equilibrium and that they are not controlled by density-dependent variables. In addition, species with unpredictable demography may either be out of equilibrium or influenced by density-dependent factors, such as masting events. The utility of the model is that whether it correctly or incorrectly predicts tree distribution patterns, the results allow users to understand more about which variables may or may not be influencing demography patterns.

INTRODUCTION

Growth, mortality, and tree regeneration drive the major patterns in forest demography (Wunder 2007). Quantifying and understanding the fundamental relationship between these variables can inform historical forest dynamics and help predict future changes. Many theoretical and empirical studies have addressed how these variables independently affect forests. In one theoretical study, Chesson (1998) concluded that most populations should be considered at least partly limited by recruitment. In another, artificially thinning a red pine forest significantly altered forest dynamics by increasing the mortality rates (Powers et al. 2010). Growth has also been shown to be important for forest dynamics as a growth strategy trade-off (Howard and Goldberg 2001) and for its role in density dependence (e.g. Boiven 2010). Unfortunately, fewer theoretical models have addressed all three variables in a testable model that applies to trees.

In 2007, a model by Artzy-Randrup et al. applied growth, mortality, and recruitment to coral demographics. Although this model focused on corals, it can also be applied to tree demography. Both planktonic larvae of coral and tree fruit migrate within and among nodes in an open system. Similar to coral, trees become sessile upon germination and then grow steadily for most of their life. By assuming a size-dependent growth pattern for each species, Artzy-Randrup et al. (2007) used population size structure as a proxy for changes in population through time. Although this invokes several assumptions, it enables predictions of population dynamics from a single survey.

The purpose of our study was to use the Artzy-Randrup et al. (2007) model to determine demographic information about the 16 hectares of deciduous trees (>10 cm GBH) located in the E.S. George Reserve (data collected by Allen). Our study’s motivating questions were: 1) how do the empirical size-structures of individual species correlate with those predicted by the model,
2) why or why not might this be the case, and 3) can a model based on coral populations help us understand forest dynamics?

**METHODS**

*Adapted Demography Model*

Our adaptations of the Artzy-Randrup et al. (2007) model are such:

Let \( n(x,t) \) be the size distribution of the population at time \( t \). The general transport equation, which holds regardless of assumptions, is:

\[
\partial_t n(x, t) + \partial_x (g(x, t)n(x, t)) = -u(x, t)n(x, t)
\]

where \( g(x,t) \) is the rate at which individuals of size \( x \) at time \( t \) grow to larger sizes, and \( u(x,t) \) is the individual death rate. For a demonstration of this equation, see (Artzy-Randrup et al. 2007).

For our system, we assume a constant death rate, independent of size and density, \( u(x,t) = u \), and a density-independent individual growth function, \( g(x,t) = g(x) \). The equation then becomes

\[
\frac{d}{dx} (g(x)n(x)) = -un(x)
\]

To solve it, we write \( g(x)n(x) = Ke^{a(x)} \), where \( K \) is some arbitrary normalizing constant and \( a \) is some function of \( x \). Substituting it back in the equation gives

\[
\frac{da}{dx} = -\frac{u}{g(x)}
\]

whose solution is

\[
a(x) = -u \int_{x_0}^x dx/g(x)
\]

where arbitrary constants were set to zero without loss of generality (they ultimately appear in the normalizing factor \( K \)). The growth function is constructed based on biological assumptions. We will assume \( g(x) = g_1(x)g_2(x) \), where \( g_1 \) dominates at small size \( (x \approx x_0, \text{ where } x_0 \text{ is the size at which new individuals enter the system}) \), and \( g_2 \) dominates at big size \( (x \approx x_{max}, \text{ where } x_{max} \text{ is the maximum size individuals can possibly reach, which in our case could be the size of the largest tree in the Big Woods}) \). Assuming radial growth occurs at a constant rate when the tree is small and free to grow, we have \( g_1 = \frac{dx}{dt} = \lambda \). On the other hand, we assume growth stalls near \( x = x_{max} \). A general form for \( g_2 \) is thus \( g_2 = (1 - x/x_{max})^d \). The parameter \( d \) (taken to be \( > 1 \)) is a measure of how fast \( g_2 \) drops to zero as \( x \rightarrow x_{max} \) (Figure 1). We can then solve for \( a(x) \) to complete our model, which is then comprised of the following expressions:

\[
n(x) = Ke^{a(x)}
\]

\[
g(x) = \lambda(1 - x/x_{max})^d
\]
\[ a(x) = \frac{u}{\lambda(1 - d)} x^d \left( \frac{1}{x_{\text{max}} - x} - \frac{1}{x_{\text{max}} - x_0} \right) \]

**Empirical Data Analysis**

We used Matlab and R to extract demography patterns of the major tree species in the Big Woods. Analyzed species included *Prunus serotina*, *Acer rubrum*, *Hamamelis virginiana*, *Quercus alba*, *Quercus rubra/veluntina*, and *Carya spp.* We then tested our model against these species by comparing the curves the model can describe (depending on the parameters \( u, \lambda, d \)) with our empirical curves. Our interpretation is that if a given species curve does not match any curve the model can produce, then the species is not in equilibrium or there is some density-dependence at play. However, if the species does fit a possible model curve, we can assume our assumptions hold. By then estimating the parameters that apply to that species, we can interpret its growth patterns. In either case, the test can in principle suggest whether or not the species is shade-tolerant, fast growing, and recruitment limited\(^1\).

**RESULTS**

The distribution of diameters for adult *Prunus serotina*, *Acer rubrum*, and *Hamamelis virginiana* follow a similar distribution pattern (Figures 1-3). There is a high frequency of trees with small diameters followed by a sharp drop in the number of trees with larger diameters. As can be seen from Figure 4, this distribution can be produced by our model when specific combinations of parameters are used.

---

\(^1\) Recruitment limitation is defined here as follows: a recruit-limited species does not suffer especially high mortality at its early life stages; instead, most seedlings make it to adult size. For such species, the number of new individuals is what limits its population size. A tree that is not recruit-limited, however, suffers high mortality before reaching safety at adult size (when it reaches the canopy and is thus free of deer browsing and/or competition for light). This kind is more limited by free space than by number of recruits.
Figure 3: Distribution of *Hamamelis virginiana* diameters in the Big Woods

Figure 4: Model-produced distribution of size classes ($\lambda = 0.4$ and $u = 0.4$).

*Quercus alba* and *Quercus veluntina/rubra* show a qualitatively different distribution. They show a bimodal distribution, with the first peak at a young stage and the second at adult size. *Carya spp.* show a similar trend, though the overall pattern is shifted to later life stages. When these distributions are compared with Figures 4 and 8, we see that our model can describe either peak but not the bimodal distribution.

Figure 5: Distribution of *Quercus alba* diameters in the Big Woods

Figure 6: Distribution of *Quercus veluntina/rubra* diameters in the Big Woods
DISCUSSION

The model is inherently simple so that it can capture the essence of the most basic forest dynamics. In an effort to preserve simplicity, we kept the death rate in the model constant across size classes. By changing $\lambda$ and $d$, our model provided us with comparable curves for some tree species diameter distributions but not for others.

The distributions for *Prunus serotina*, *Acer rubrum* and *Hamamelis virginiana* can be successfully modeled by shifting $\lambda$ and $d$. Given this match, we cannot in principle rule out the possibility that the assumptions made in the model hold for these trees. If that is the case, then our trees are in a state of equilibrium and there are no density-dependent factors acting on them. In other words, the pattern we see could imply that these trees are not subject to biotic interactions such as competition. Instead, their abundances are limited solely by the number of recruits and their demography by the mechanical processes of growth and death. However, the Big Woods previously underwent periodic forest fires until approximately 150 years ago (Allen, personal communication, 2010) and should be considered a system that has not reached equilibrium yet. The fires previously suppressed *Acer spp.* and *Prunus spp.* seedlings from growing, which resulted in an oak-hickory dominated canopy. When the fires were no longer set, more *Acer spp.* and *Prunus spp.* seedlings could survive in the forest, mimicking a high density seedling event. This would explain the abundance of small-sized *Acer spp.* and *Prunus spp.* We would then expect the observed peaks to shift to larger diameters as time continues.

The bimodal histograms observed in *Quercus spp.* and *Carya spp.* cannot be replicated by our model. The discrepancy between the empirical and model curves implies that our assumptions have been violated. Therefore, either these trees are out of equilibrium or their demography is informed by density-dependent factors. Since we know that the Big Woods is undergoing a transition in species composition, we can interpret our findings under a non-equilibrium situation. The abundance of large-sized trees may be a result of seedlings that survived the previous fire regime and grew to adulthood. A combination of mast years and
competition with *Acer spp.* and *Prunus spp.* seedlings could explain the remaining bimodal pattern.

**CONCLUSIONS**

This study shows how a very simple and arguably unrealistic model can provide insights into interesting questions about our study system. We learn from its success that further complications might just not be necessary to explain the demography of some trees. From its failure and in conjunction with historical knowledge about the Big Woods, we learn that some tree species have not yet reached equilibrium. This information can now help inform future model development. For instance, future models may consider incorporating more sophisticated assumptions about disturbances, growth functions and death rates.

**ACKNOWLEDGEMENTS**

Many thanks to Dave Allen for access to his valuable dataset on the Big Woods. Thanks to Yael Artzy-Randrup, for inadvertently contributing to the only model-focused field problem in the Field Ecology 2010 season.

**LITERATURE CITED**


ABSTRACT

The introduction of non-indigenous species to marine ecosystems is increasing rapidly due to anthropogenic causes. The most recent wave of introduced species in Lake Erie included the round goby (Neogobius melanostomus) and the tubenose goby (Proterorhinus semilunaris). These two non-indigenous species are now well-established in the Great Lakes, with the round goby becoming one of the most common benthic nearshore fish. These species have been documented to be detrimental food competitors for other small benthic species as well as successful predators of native juvenile fish eggs. Diet habits of these goby species show that native mussel species may also be at risk due to predation. Gut contents of nearshore fish species in L. Erie were identified and analyzed in order to determine the contribution of these gobies to the recent displacement of native species due to competition for food resources. This study found no significant difference in the weight of food content for these species (p>0.05). Results from the stomach contents showed that the tube nose goby (Proterorhinus marmoratus) and the round nose goby (Neogobius melanostomus) were eating the same prey items as native benthic fishes found in Lake Erie. Significant results (Figures 1 & 2) were found for this diet overlap in the tube nose goby (Proterorhinus marmoratus) and: the black gill, black bull head, and tadpole madtom.

INTRODUCTION

Human activity has facilitated the spread of species and accelerated the rate of introduction of non-indigenous species into new environments. The introduction of a non-native species in an ecosystem represents an ecological risk if the species is able to integrate itself successfully into the ecosystem (Gertzen et al. 2008), because it can affect native biodiversity through predation, competition, hybridization, habitat modification and transmission of exotic diseases (Gozlan et al. 2010).

The Great Lakes have experienced two major waves of species introductions: the first arrived during the 1920s-1930s, and consisted of rainbow smelt (Osmerus mordax), alewife (Alosa pseudoharengus), and sea lamprey (Petromyzon marinus). The second, more recent wave, entered via ballast water from ocean-going ships, and included the spiny water flea (Bythotrephes longimanus), dreissenid mussels (D. polymorpha and D. bugensis), the fishhook water flea (Cercopagis pengoi), the round goby (Neogobius melanostomus) and the tubenose goby (Proterorhinus semilunaris) (Dobiesz et al. 2005).
The round goby and the tubenose goby are originally from the Ponto-Caspian basin and now are well-established in the Great Lakes, with the round goby becoming one of the most common benthic nearshore fish (Kvach and Stepien 2008). These gobies can out-compete native benthivorous fish species such as logperch (*Percina caprodes*) and mottled sculpin (*Cottus bairdii*), and similar consequences can be expected for other small benthic species. Gobies also may have direct impacts through predation on juvenile fish and fish eggs. Dreissenid mussels have been reported as the most important prey item for large round goby, which suggests that native mussel species may also be at risk due to predation. Lastly, the impacts on benthic fishes, may have indirect effects for unionid mussels, which require fish hosts to transform and disperse their glochidia larvae (Poose et al. 2010).

In this study we examined the gut contents of nearshore fish species in Lake Erie, in order to determine if the invasive gobies are displacing native species due to competition for food resources.

**METHODS AND MATERIALS**

*Sampling Methodology*

On October 16, 2006 we surveyed and collected specimens from a boat port at the mouth of the Huron River, leading into Lake Erie. Sampling was done by seining with two different size nets. We began seining on the edge of the boat launch at around 10:30 AM and finished around 11:00 AM. The weather conditions were sunny and clear. The waters were not turbulent and thus marine activity was at normal level. The wind speed was low, between 0 and 5 mph. Water temperature was measured with a temperature probe and found to be at 4°C. All species of gobies were collected as well as six other native benthic fish species including: *Salvelinus fontinalis* (brook trout), *Ambloplites rupestris* (rock bass), *Lepomis gibbosus* (pumpkinseed), *Noturus gyrinus* (tadpole madtom), *Lepomis macrochirus* (bluegill), and *Ameiurus melas* (black bullhead); all others were returned to the lake. All fish were transported back to the lab in an ice cooler.

*Laboratory Technique*

Measurements of total length and sex determination were conducted for each sample. Stomach along with intestinal contents were extracted and analyzed by cutting the fish open from mouth to anus. The stomach, and its contents, was weighed for each individual. Dissecting microscopes were used to identify and estimate the relative percentage of each food type found in the guts.

**RESULTS**

There was no significant difference in the weight of food content for these species (p>0.05). Most of the fish were found to have food in their stomach. Round goby, black bullhead and pumpkin seed had more food (Table 1) than the others, however, they showed to have a different diet according to the percentage found from the prey item analysis.
Table 1: Food content for different Species

<table>
<thead>
<tr>
<th>Species</th>
<th>Mean of weight of contents</th>
<th>% with food</th>
</tr>
</thead>
<tbody>
<tr>
<td>black bullhead</td>
<td>0.076±0.055</td>
<td>0.8</td>
</tr>
<tr>
<td>blue gill</td>
<td>0.036±0.011</td>
<td>1</td>
</tr>
<tr>
<td>tadpole madtom</td>
<td>0.013±0.005</td>
<td>1</td>
</tr>
<tr>
<td>pumpkin seed</td>
<td>0.080±0.131</td>
<td>0.8</td>
</tr>
<tr>
<td>rock bass</td>
<td>0.033±0.037</td>
<td>0.83</td>
</tr>
<tr>
<td>round goby</td>
<td>0.076±0.157</td>
<td>0.86</td>
</tr>
<tr>
<td>brook trout</td>
<td>0.008±0.004</td>
<td>0.8</td>
</tr>
<tr>
<td>tubenose goby</td>
<td>0.015±0.012</td>
<td>1</td>
</tr>
</tbody>
</table>

The round goby, black bullhead, rock bass, brook trout, pumpkin seed, blue gill, and tadpole madtom did not show any correlation between body length and weight of content. Body length is closely connected to body weight, thus this result further suggests that the body weight did not influence predation for those species. But TN (tube nose goby) showed significant correlation ($R^2=0.626$, $p=0.03$) suggesting bigger TN would be more competitive than smaller individuals of this species.

Tubenose goby, blue gill, black bullhead, brook trout mainly depend on chiromonidae (CN) as their major food resource. The proportion of chiromonidae (CN) in the diet of the tube nose goby was the largest among these species, which suggests that the tube nose goby could be a crucial competitor for the native species (Figures 1 & 2).

Figure 1. Comparison of diet for different species.
DISCUSSION

There is a growing body of evidence that the introduction of non-indigenous species to marine ecosystems is increasing rapidly due to global transport (Jude et al. 1992, Carlton 1996, Ruiz et al. 1997). As a consequence, marine and estuarine habitats have experienced dramatic ecosystem and trophic levels alterations. Introduced species not only prey on native species, but may also compete with them for food resources and spawning habitats (Lodge 1993, Ruiz et al. 1997, Karlson et al. 2007). Gobies have several biological attributes that facilitate their successful invasion. Gobies exhibit phenotypic plasticity (Kováč and Siryová 2005, L’avrinčikova et al. 2005), effective reproductive tactics (Growbaska 2005), and generally have low parasite loads compared to native species (Ondračková et al. 2005). Assessment of the gut contents is a commonly used technique to assess diet preferences and identify trophic relationships in fish (Jude et al. 1987, Karlson et al. 2007, Pilger et al. 2010).

Results from the stomach contents showed that the tube nose goby (Proterorhinus marmoratus) and the round nose goby (Neogobius melanostomus) were eating the same prey items such as chironomids, mayfly, and ostracods. Interestingly, all of the native species also ate chironomids, with others exhibiting a higher percentage of chironomid dietary preference similar to P. marmoratus. This suggests an overlap in dietary preferences between native and introduced species. This adds to the growing body of evidence signifying that gobies are generally opportunistic feeders (Grabowska and Grabowski 2005; Kakareko et al. 2005). However, approximately 60% of the diet of P. marmoratus consists primarily of chironomids (See Figure x). Although the Bluegill (Lepomis macrochirus macrochirus) and Black bullhead (Ameiurus melas) had gut contents consisting of higher percentages of chironomidae (> 50%, See Figure x)
compared to all other native species and including *N. melanostomus, P. marmoratus* was still the highest consumer of chironomid prey. This suggests a competitive interaction between *P. marmoratus* and the other native species.

Additionally, with the exception of *P. marmoratus*, the relationship between the total length of the fish and the weight of the stomach contents was not significant. At least for *P. marmoratus*, its competitive advantage for food resources is dependent on its size and age. The size of *P. marmoratus* may be directly proportional to the amount and size of the prey it consumes. An ontogeny study by Crisp and Carrick (1975) provided evidence for the direct relationship between total length and age in fish. They concluded that the total length of the fish increases as the fish matures. If this same condition applies to *P. marmoratus*, then it is possible that its competitive advantage is also due to its ability to find food over its lifetime. Time series experiments conducted during the times at which each species feed would further elucidate diet preferences between the native and introduced species. Furthermore, prey preference between the native and introduced species would also be elucidated in laboratory experiments mimicking natural conditions (i.e. temperature, salinity, habitat characteristics). Fish species would be starved and acclimatized to the natural conditions set up in the lab before observing their feeding strategies.

ACKNOWLEDGEMENTS

We would like to thank Dave Allen, Ivette Perfecto, and John Vandermeer.

LITERATURE CITED


Abstract: Mary Talbot’s extensive survey of the ant species of the E.S. George Reserve offers an opportunity to examine changes in species diversity over time. We compared the species present on Bee Island from Talbot’s surveys to those present today in order to understand how ant diversity had changed over 60 years. We collected ants via numerous methods in order to conduct a more thorough sampling effort and obtain a maximum number of species. Several species identified by Talbot and collaborators were found on the island, but a number of species were identified that were novel to Bee Island in the time since the original surveys.

INTRODUCTION

Ants are classified into a single family, Formicidae, in the order Hymenoptera, which also contains bees, wasps, and sawflies (Hölldobler & Wilson, 1990). In 1990, Hölldobler and Wilson stated there were 11 subfamilies, 297 genera, and approximately 8,800 species of described ants, and possibly over 20,000 undescribed species. In addition to their immense diversity, the sheer biomass of ants makes them important members of various ecosystems. Ants and termites alone can comprise up to one-third of the biomass of a hectare of tropical forest (Wilson, 1987). Ants also play important roles in structuring communities through their roles as predators, pollinators, and seed dispersers. Ants are involved in nutrient cycling in the soil and often develop symbiotic relationships with animal species, thus the activity of ants and the distribution of their colonies influence fauna and flora communities (Roth 1994). For these reasons it is easy to understand why E.O. Wilson described these invertebrates as “the little things that run the world” (Wilson, 1987).

Despite the obvious prevalence and importance of ants, we can only speculate as to why ants and other invertebrate species are so diverse. One of the greatest questions in ecology is why and how extensive biodiversity is able to persist in nature. One prediction states that the small size of ants and their corresponding niches might be a key reason (Wilson, 1987). The prediction follows that ants can divide up the environment into many little domains where specialists can coexist. Another explanation is the idea that diversity at one level leads to diversity at other levels. This hypothesis is based on the assumption that consumers are not equally fit on all resource types. Therefore we anticipate a degree of niche differentiation, which consequents a reduction in interspecific competition (Ambrecht et al, 2004). Furthermore, the dispersion of ants may be linked to outcomes of competition, variations in soil, topography, or land use practices. Behavioral interactions can affect the spacing of ant colonies within a local neighborhood, but abiotic factors influence population distribution and community interactions at broader scales (Crist & Wiens, 1996).

Previous studies have shown that changes in soils associated with land use can alter distributional ranges of ants. Due to their wide distribution throughout the world in diverse habitats, ants have the potential to serve as strong indicators of biological diversity. Degrees of habitat disturbance or succession in a community can often be reflected in its ant assemblages.
(Roth & Perfecto, 1994). Our study attempted to investigate the ways in which changes in land use practices at the E.S. George Reserve has affected ant diversity. In the 1950s Mary Talbot began the first comprehensive study of Formicidae in North America. Her study was conducted at the E. S. George Reserve and resulted in the identification of 87 total species (Talbot, 1975). Talbot recorded the location of each species found on the Reserve and found a total of 25 species on Bee Island and the surrounding swamp. Our study resurveyed Bee Island to determine the changes in ant community assemblage over time.

We anticipated finding overlap of species between the two surveys, as well as new species that have recently colonized the island and the corresponding displacement of species previously found by Talbot. The E.S. George Reserve has experienced the invasion of new species of ants since the 1950s and undergone many changes to the forest structure. Experiments with deer population management and reforestation of agricultural areas have predictably led to differences in community structures of sites in the reserve. The collection and identification of ant species on Bee Island could elucidate environmental changes on the E. S. George Reserve and how these changes have affected biodiversity.

MATERIALS AND METHODS

A survey of all ant species observed was performed on Bee Island, a small island just east of Big Island in the Big Swamp of the ESGR. Before surveying, a copy of Mary Talbot’s study was acquired from the University of Michigan Natural History Museum and examined for all species found on and in the swamp around Bee Island in the 1950’s. All species present were recorded for future reference.

Collection

Three survey methods were used to acquire ants: baiting, hand collecting, and litter sifting. All collected ants were placed in one vial of 70% ethanol per occurrence (either a singular bait, singular ant, or singular area), marked with its own accession number designated, which coordinated with notes taken elsewhere on location and activity of the ant(s) collected.

Baiting

Two parallel 30m transects were laid across a level area of the island (Fig. 1), and alternating salt water-soaked and sugar water-soaked cotton balls were placed every meter along one transect (Transect 1). Tuna was placed every two meters on the second transect (Transect 2), with each placement halfway between a salt and sugar bait. Baits were checked periodically for the presence of ants and all ants present were collected.
Hand collecting

Team members who hand collected ants targeted areas where ants were most likely to be found (by rotting logs, in large amounts of leaf litter). Hand collected ants were acquired via aspirators or soft forceps.

Litter Sifting

Team members who litter sifted first picked a 1m² area of leaf litter likely to have ants, chopped the area with a trowel to stir up ants from the ground, and then transferred all leaf litter into a sifting bag. After sifting for two minutes, the larger litter was dumped out of the top compartment of the bag, while the smaller litter and organisms were trapped in a lower bag. This procedure was repeated 10 times, and all collected litter was collected into a single bag. Any ants seen while litter sifting were immediately collected. All small litter was placed in a Winkler trap to dry for 28 hours, and fallen ants were collected in bags of 70% ethanol.

All three methods were performed once in the morning and revisited in the afternoon. The baiting transects were re-examined for presence of ants, and those performing hand collections also laid a new transect (Transect 3) of tuna bait which they examined for the presence of ants. Litter sifting in the afternoon repeated the morning’s procedures with 10 new 1m² areas, and a new Winkler trap was set to dry for 24 hours.

Pinning and Identification

Once all collection surveys had concluded, one representative individual of each ant species from each vial was pointed and pinned for examination, and each ant was accompanied by an accession number from the vial to which it was first collected. Each pinned ant was inspected via microscope and keyed to subfamily, genus, and species (or morpho-species) using online resources such as AntWeb and the Global Ant Project, as well as published keys, including *Ants of Ohio* and *Ants of North America*. A list of all identified representative individuals, their accession number, their subfamily, their genus, and their species was compiled.

RESULTS

Talbot documented 25 ant species from Bee Island in 1950 (see Appendix, Table 1), our survey yielded similar diversity. We encountered 22 ant species on Bee Island in the sub-families Dolichoderinae, Formicinae, Ponerinae, and Myrmicinae (see Appendix, Table 2). We were not able to identify all specimens to species. Many species in the genera *Formica* and *Myrmica* were marked as morpho-species. Assuming that three of our unidentified *Formica* spp. overlapped with Talbot’s *Formica* spp., we observe 11 species of overlap between studies, or approximately 44% of all species Talbot described from the island (Talbot, 1975). If we exclude morpho-species from the list of species we found on the island, but that Talbot did not find, we observe 3 species (*Temnothorax longispinosis*, *Myrmica latifrons*, and *Myrmica lobifrons*) that were not...
recorded in Talbot’s survey of the entire E. S. George Reserve. Changes in taxonomy since Talbot’s surveys were taken into account during identification.

DISCUSSION

The contrasting biodiversity in ant species found by Talbot and our recent survey of one site on the E. S. George Reserve has important implications in understanding the temporal changes that have occurred between the two studies, particularly when examining the biodiversity of invertebrates and other fauna found in our community (Anderson, 2004). The study site as the basis of comparison was of particular interest to us. Species richness should remain consistent within a habitat and as expected, the total number of species remained approximately the same. What we could not have predicted is that the variability of the species present would be so high when making comparisons between the two surveys. Speculation as to what may explain the differences should all be prefaced with the constraints that although we have documentation and data confirming the species found by Talbot, there were no records kept of methodologies and sampling procedures (Talbot, 1975). Within our study, we were able to see that different techniques can result in entirely different censuses obtained. The variation between the baiting and sifting methods compared to hand collecting and using an aspirator for collecting specimens can result in a fewer number of individuals but more biodiversity for the latter, as you maintain the flexibility to move between microhabitats. A more precise comparison could be made if the exact procedure used by Talbot was available. Furthermore, it isn’t known how extensively Talbot surveyed Bee Island, in which case our survey may have been more thorough, explaining how we obtained greater diversity than Talbot.

One major difference between our survey and the Talbot survey is the presence of the Winter Ant, *Prenolepis imparis*. *P. imparis* constituted the highest proportion of all of the ants specimens collected. However, this species in particular was not present in the previous study, and Talbot's expertise would indicate a low likelihood of it being overlooked. It’s likely this discrepancy in sampling is due to the time of year that the surveys were conducted; our survey took place on a cold fall day when most ant species are not active (with the exception of *P. imparis*), whereas Talbot likely sampled the site during the summer. Additionally *A. fulva*, *T. longispinosis*, *L. umbratus*, and *S. schmitti* all are now found seemingly at the northern most limits of their ranges. Movement along the latitudinal gradient has long been associated with climatic changes forcing species to migrate north to evade hotter temperatures or now being able to access previously unsuitable regions due to the harsher winters/colder climate. Typically the first sign of this mechanism can be witnessed within invertebrate communities, as they are more responsive to change in abiotic factors such as temperature (Kremen 1993) The flux of new species into the community can have untold consequences, presumably local extinctions of species that we were unable to find from the first census. Because of the vast number of biotic and abiotic factors that could contribute to our findings, we recommend further survey and broader analyses of the variation on the E. S. George Reserve to determine if our findings are reflective of a broader change in the system such as climatic changes or more cyclical patterns within the populations (Didham 2003). Future surveys should aim to not only sample the ants of Bee Island, but also other locations with the E.S. George Reserve.

Acknowledgements
Many thanks to Ivette Perfecto for the use of her laboratory during identification proceedings.
LITERATURE


APPENDIX

Table 1. List of species documented in the E. S. George Reserve in Talbot (1975).
<table>
<thead>
<tr>
<th>Sub-family</th>
<th>Sub-family</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Dolichoderinae</strong></td>
<td><strong>Formicinae continued</strong></td>
</tr>
<tr>
<td>Dolichoderus mariae Forel</td>
<td>Formica ulkei Emery</td>
</tr>
<tr>
<td>Dolichoderus plagiatius (Mayr)</td>
<td>Formica vinculans Wheeler</td>
</tr>
<tr>
<td>Dolichoderus pustulatus Mayr</td>
<td>Formica sp.</td>
</tr>
<tr>
<td>Dolichoderus taschenbergi (Mayr)</td>
<td>Lasius alienus (Foerster)</td>
</tr>
<tr>
<td>Dorymyrmex pyramicus (Roger)</td>
<td>Lasius flavus (Fabricius)</td>
</tr>
<tr>
<td>Tapinoma sessile (Say)</td>
<td>Lasius minutus Emery</td>
</tr>
<tr>
<td></td>
<td>Lasius nearcticus Wheeler</td>
</tr>
<tr>
<td></td>
<td>Lasius neoniger Emery</td>
</tr>
<tr>
<td></td>
<td>Lasius pallitarsis (Provancher)</td>
</tr>
<tr>
<td><strong>Formicinae</strong></td>
<td></td>
</tr>
<tr>
<td>Acanthomyops claviger (Roger)</td>
<td>Lasius speculiventris Emery</td>
</tr>
<tr>
<td>Acanthomyops interjectus (Mayr)</td>
<td>Lasius umbratus (Nylander)</td>
</tr>
<tr>
<td>Acanthomyops latipes (Walsh)</td>
<td>Paratrechina parvula (Mayr)</td>
</tr>
<tr>
<td>Acanthomyops murphyi (Forel)</td>
<td>Polyergus lucidus Mayr</td>
</tr>
<tr>
<td>Acanthomyops subglaber (Emery)</td>
<td>Prenolepis imparis (Say)</td>
</tr>
<tr>
<td>Brachymyrnex depilis Emery</td>
<td>Lasius speculiventris Emery</td>
</tr>
<tr>
<td>Camponotus americanus Mayr</td>
<td>Lasius umbratus (Nylander)</td>
</tr>
<tr>
<td>Camponotus caryae (Fitch)</td>
<td>Paratrechina parvula (Mayr)</td>
</tr>
<tr>
<td>Camponotus nearcticus Emery</td>
<td>Polyergus lucidus Mayr</td>
</tr>
<tr>
<td>Camponotus novaeboracensis (Fitch)</td>
<td>Prenolepis imparis (Say)</td>
</tr>
<tr>
<td>Camponotus pennsylvanica (DeGeer)</td>
<td></td>
</tr>
<tr>
<td>Formica creightoni Buren</td>
<td></td>
</tr>
<tr>
<td>Formica dakotensis Emery</td>
<td></td>
</tr>
<tr>
<td>Formica exsectoides Forel</td>
<td></td>
</tr>
<tr>
<td>Formica fusca Linnaeus</td>
<td></td>
</tr>
<tr>
<td>Formica glacialis Wheeler</td>
<td></td>
</tr>
<tr>
<td>Formica lasioides Emery</td>
<td></td>
</tr>
<tr>
<td>Formica neogagtes Emery</td>
<td></td>
</tr>
<tr>
<td>Formica neorufibaris Emery</td>
<td></td>
</tr>
<tr>
<td>Formica nepticula Wheeler</td>
<td></td>
</tr>
<tr>
<td>Formica obscuroipes Forel</td>
<td></td>
</tr>
<tr>
<td>Formica obscuriventris Mayr</td>
<td></td>
</tr>
<tr>
<td>Formica pallidefulva nitidiventris Emery</td>
<td>Crematogaster cerasi Fitch</td>
</tr>
<tr>
<td>Formica pergandei Emery</td>
<td></td>
</tr>
<tr>
<td>Formica rubicunda Emery</td>
<td></td>
</tr>
<tr>
<td>Formica shauffusi Mayr</td>
<td></td>
</tr>
<tr>
<td>Formica vinculans Wheeler</td>
<td></td>
</tr>
<tr>
<td>Formica subintegra Emery</td>
<td></td>
</tr>
<tr>
<td>Formica subnuda Emery</td>
<td></td>
</tr>
<tr>
<td>Formica subsericea Say</td>
<td></td>
</tr>
<tr>
<td>Formica talbotae Wilson</td>
<td></td>
</tr>
</tbody>
</table>

Table 1 continued.

Ponerinae

Amblyopone pallipes (Haldeman)
Poner a pennsylvanica Buckley
Proceratium silaceum Roger

Myrmicinae

Aphaenogaster fulva Roger
Aphaenogaster rudis (Emery)
Aphaenogaster tennesseensis (Mayr)
Aphaenogaster treatae Forel
Crematogaster cerasi Fitch
Crematogaster lineolata (Say)
Leptothorax ambiguus Emery
Leptothorax curvispinosus Mayr
Leptothorax duloticus (Wesson)
Leptothorax longispinosus Roger
Leptothorax muscorum (Nylander)
Leptothorax shaumi Roger
Leptothorax texanus Wheeler

<p>| 187 |</p>
<table>
<thead>
<tr>
<th>Sub-family</th>
<th>Sub-family</th>
</tr>
</thead>
<tbody>
<tr>
<td>Myrmicinae continued</td>
<td>Myrmicinae continued</td>
</tr>
<tr>
<td><em>Leptothorax hirticornis</em> Emery</td>
<td><em>Myrmica pinetorum</em> Wheeler</td>
</tr>
<tr>
<td><em>Monomorium minimum</em> (Buckley)</td>
<td><em>Myrmica punctiventris</em> Roger</td>
</tr>
<tr>
<td><em>Monomorium talbotae</em> DuBois</td>
<td><em>Solenopsis molesta</em> (Say)</td>
</tr>
<tr>
<td><em>Myrmecina americana</em> Emery</td>
<td><em>Stenamma brevicorne</em> (Mayr)</td>
</tr>
<tr>
<td><em>Myrmica americana</em> Weber</td>
<td><em>Stenamma diecki</em> Emery</td>
</tr>
<tr>
<td><em>Myrmica discontinua</em> Emery</td>
<td><em>Stenamma impar</em> Forel</td>
</tr>
<tr>
<td><em>Myrmica emeryana</em> Forel</td>
<td><em>Stenamma schmitii</em> Wheeler</td>
</tr>
<tr>
<td><em>Myrmica fracticornis</em> Emery</td>
<td><em>Smithistruma</em> (Pyramica) pergandei (Emery)</td>
</tr>
<tr>
<td><em>Myrmica incompleta</em> Provancher</td>
<td><em>Smithistruma</em> (Pyramica) pulchella (Emery)</td>
</tr>
<tr>
<td><em>Myrmica spatulata</em> M. R. Smith</td>
<td></td>
</tr>
</tbody>
</table>

Table 2. Ant species described from Bee Island in 1950 by Talbot and the current study.
<table>
<thead>
<tr>
<th><strong>Aphaenogaster rudis</strong></th>
<th><strong>Aphaenogaster rudis</strong></th>
<th><strong>Aphaenogaster fulva</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Aphaenogaster tennesseensis</strong></td>
<td></td>
<td><strong>Formica sp. 4</strong></td>
</tr>
<tr>
<td><strong>Camponotus nearcticus</strong></td>
<td><strong>Camponotus nearcticus</strong></td>
<td><strong>Temnothorax longispinosus</strong></td>
</tr>
<tr>
<td><strong>Camponotus noveboracensis</strong></td>
<td></td>
<td><strong>Myrmica detritinodis</strong></td>
</tr>
<tr>
<td><strong>Camponotus pennslyvanicus</strong></td>
<td><strong>Camponotus pennslyvanicus</strong></td>
<td><strong>Myrmica sp. 1 c.f. latifrons</strong></td>
</tr>
<tr>
<td><strong>Cremaogaster cerasi</strong></td>
<td></td>
<td><strong>Myrmica sp. 2 c.f. lobifrons</strong></td>
</tr>
<tr>
<td><strong>Cremaogaster lineolata</strong></td>
<td></td>
<td><strong>Myrmica sp. 3</strong></td>
</tr>
<tr>
<td><strong>Dolichoderus mariae</strong></td>
<td></td>
<td><strong>Myrmica sp. 4</strong></td>
</tr>
<tr>
<td><strong>Dolichoderus plagiatus</strong></td>
<td></td>
<td><strong>Lasius umbratus</strong></td>
</tr>
<tr>
<td><strong>Formica neogagates</strong></td>
<td><strong>Formica sp. 1?</strong></td>
<td><strong>Prenolepis imparis</strong></td>
</tr>
<tr>
<td><strong>Formica obscuriventris</strong></td>
<td><strong>Formica sp. 2?</strong></td>
<td><strong>Stenamma schmitti</strong></td>
</tr>
<tr>
<td><strong>Formica subintegra</strong></td>
<td><strong>Formica sp. 3?</strong></td>
<td></td>
</tr>
<tr>
<td><strong>Lasius alienus</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Lasius minutus</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Lasius pallitarsis</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Leptothorax ambiguus</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Leptothorax curvispinosus</strong></td>
<td><strong>Leptothorax curvispinosus</strong></td>
<td></td>
</tr>
<tr>
<td><strong>Leptothorax longispinosus</strong></td>
<td><strong>Leptothorax longispinosus</strong></td>
<td></td>
</tr>
<tr>
<td><strong>Leptothorax muscorum</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Leptothorax schaumi</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Myrmica americana</strong></td>
<td><strong>Myrmica americana</strong></td>
<td></td>
</tr>
<tr>
<td><strong>Ponera pennslyvanica</strong></td>
<td><strong>Ponera pennslyvanica</strong></td>
<td></td>
</tr>
<tr>
<td><strong>Solenopsis molesta</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Stenamma diecki</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Tapinoma sessile</strong></td>
<td><strong>Tapinoma sessile</strong></td>
<td></td>
</tr>
</tbody>
</table>
A RE-EXAMINATION OF PEROMYSCUS LEUCOPUS DEMOGRAPHY AND SOURCE-SINK DYNAMICS ON THE E. S. GEORGE RESERVE

CINDY BICK

ABSTRACT

*Peromyscus leucopus* is one of the most abundant forest dwelling mammals in North America. Small mammal surveys on the E. S. George Reserve continually show higher *P. leucopus* abundance compared to other small mammal present in the area. To re-examine whether *P. leucopus* populations follows the source-sink model as well as assess periodic trends of rise and fall in abundance, live trapping techniques were used. Sex, age, and reproductive status were recorded for all species trapped. Based on a small sample size, *Peromyscus leucopus* juvenile males were mostly found in the field compared to the forest. Trapping success was very low compared to the study conducted on September 25th by Chara et al. (2010). Assessment of trapping success over a four year period showed a periodic rise and fall in abundance in *P. leucopus* populations.

INTRODUCTION

The white-footed mouse (*Peromyscus leucopus*) is a common forest dwelling mammal inhabiting most of eastern and central North America, ranging from Nova Scotia to Virginia (Wilson and Ruff 1999). *Peromyscus leucopus* typically inhabits woodland and mixed forests in areas interspersed with herbaceous cover and abundant with rocky outcrops, stumps and logs (Kurta 1995). They are also common near agricultural lands (Wilson and Ruff 1999). In Michigan, *Peromyscus leucopus* is found at the Lower Peninsula and the southernmost Upper Peninsula (Myers et al., 2005). Currently, the distribution of *P. leucopus* has now shifted to occupy all of the Upper Peninsula except for its eastern tip (Myers et al. 2009). *Peromyscus leucopus* appears to be the most common small mammal present at the E. S. George Reserve (Journal of Field Ecology 1998-2000, 2006-2009). Its diet consists mostly of seeds and nuts from hickory, maple, oak and pine trees; therefore functioning as seed predators and dispersers (Kurta 1995). *Peromyscus leucopus* is also a food source for various predatory mammals and birds (Metzgar 1967). Climate change and habitat alteration have forced some small mammals to occupy higher latitudes (Long & Long 1993, Myers et al. 2009, Myers et al. 2005). Changes in animal distributions can have direct and indirect effects on various ecological functions as well as trophic level interactions (Junger 2004).

It has been suggested that *Peromyscus leucopus* population appears to follow a trend of periodic rise and fall in abundance (Phil Meyers, pers. comm. 2010). The availability of food resources as well as suitable habitats has been attributed to reproductive success and higher densities in *P. leucopus* (Wilder 2003). *Peromyscus leucopus* has been suggested to follow a source-sink model distribution, where the healthy reproductive adults are found in favorable habitats which act as a “source.” In contrast, the younger non-reproductive individuals are found in less favorable habitats considered the “sink” (Wilder 2003).

Forests are considered favorable habitats or sources due to the abundance of food and shelter. Open fields appear to be less favorable habitats for *P. leucopus* because there are no overstory trees such as hickory and oak, which provide not only shelter but the nuts and seeds...
that make up its main diet. Additionally, females tend to be very aggressive during breeding season and will most likely compete over prime nesting spaces (Kurta 1995). This study re-examines whether *P. leucopus* populations follow a source-sink model distribution and whether it follows a trend of periodic rise and fall in population abundance.

**METHODS**

Trapping was conducted on October 23, 2010 from 16:30-20:00 at the University of Michigan E. S. George Reserve. A hundred Sherman traps were set out along semi-linear transects in the Big Woods area, an old mature hardwood forest. The overstory is dominated by *Quercus* and *Carya* tree species, whereas the understory is dominated by *Acer rubrum*, *Prunus serotina*, and *Hammamelis virginiana*. Another hundred Sherman traps were also set out along semi-linear transects in the Air Field and Evan’s Old Field. These sites are open and grassy fields interspersed with *Juniperis communis* and *Elaeagnus umbellata*. Each trap was placed approximately 10 meters apart from each other. Traps in the forest were placed under trees and near fallen logs, while traps in the field were placed under shrubs and in tall grassy areas. A mixture containing corn, sunflower seeds and oats was placed in each trap as bait.

Traps were collected on October 24, 2010 at 09:00. Trapped individuals were identified by species and weighed to the nearest gram. The sex, age and reproductive status of each individual was also recorded. *P. leucopus* was specifically aged according to its pelage coloration and weight. Juveniles can be distinguished from the sub-adult and adult by the gray fur covering its entire dorsal area. The sub-adults have both gray and brown fur on its back. In contrast, pelage coloration in adults is brown. Males were identified as scrotal if the testes have descended and non-scrotal if the testes have not descended. A female’s reproduction condition was described as either pregnant or having tiny or enlarged nipples. All individuals were returned to the original location in which they were trapped.

**RESULTS**

As with the previous small mammal surveys conducted at the E. S. George Reserve, *P. leucopus* was the most trapped small mammal (Figure 1). Overall, only 7 individuals were trapped; 5 *P. leucopus* and 2 *Tamias striatus* (Figure 2). Of the 2 individuals of *T. striatus*, both were male with one found in the forest and the other in the field. Only 1 female *P. leucopus* was found in the field. An equal number of males were found in the forest and in the field (Figure 2). During the course of the survey, the female *P. leucopus* gave birth to three young while still within the trap. The young were only included in the total number of species distribution but were not used in the sex and age distribution analyses because they were too small to be identified as either male or female (Figure 1).

A total of 5 juvenile males and no juvenile females were trapped: 1 *T. striatus* and 4 *P. leucopus*. Localized trapping efforts showed that the total number of males in the forest and in the field were approximately the same. Specific assessment of *P. leucopus* showed the same trend (Figure 3). If the newborn young were to be included in this data, then there would be more juvenile *P. leucopus* found in the field. However, determination of which sex was most abundant in the field and the forest would be difficult to determine. Overall adult distribution showed only 1 trapped *P. leucopus* female and 1 trapped *T. striatus* male (Figure 4). The *P. leucopus* female
was lactating as she had just given birth. All of the male *P. leucopus* and *T. striatus* were non-scrotal.

Figure 1. The chart on the right illustrates the total percentage of species with identifiable age and sex that were trapped (n=7). Inclusion of the *P. leucopus* newborns are included in the chart on the left (n = 10).

Figure 2. Trapping effort only shows the total number of identifiable species (excluding the *P. leucopus* newborns) of males and females found in the forest and in the field. The females and males of all species are compared to the number of female and male *P. leucopus* caught. No females were caught in the forest.

Figure 3. Juvenile distribution of the total number of identifiable species (excluding the *P. leucopus* newborns) of males and females found in the forest and in the field. There were no juvenile females. All *P. leucopus* males were juveniles.
Small mammal trapping success at the E. S. George reserve fluctuates from year to year. Trapping success reflects the number of all individual species trapped compared to the number of traps set. The overall trapping success was compared to the trapping success of *P. leucopus*. Based on the data collected from the years 2006-2010, *P. leucopus* populations are continuously abundant in comparison to other small mammals. Their populations also seem to rise notably every 2-3 years. In 2006, the trapping success for total species as well as *P. leucopus* was the highest, followed by the lowest in 2007 (Figure 5).

Figure 5. Trapping success within the last four years reflects the total number of species trapped divided by the total number of traps set. Trapping success for *P. leucopus* is compared to the overall success. Years labeled a and b denotes trapping done at different times of the year.

DISCUSSION

Due to the limited sample size, there were difficulties in assessing whether populations of *P. leucopus* followed the distribution outlined in the source-sink model. Based on previous studies (Journal of Field Ecology 1998-2000, 2006-2009) conducted at the E. S. George Reserve,
I expected the older and reproductive individuals in the current population to be abundant in the forest; whereas the sub-adults and juveniles would be most abundant in open fields. Contrary to what was reported in the literature, there were equal numbers of *P. leucopus* juveniles in the forest and the open fields. Only one lactating female was found in the field, suggesting that perhaps *P. leucopus* populations were still in their breeding period. However, assessment of all species trapped showed the same trend found in *P. leucopus* juveniles. The juvenile males caught may have been only a few weeks old and have not been subjected to strong intraspecific competition, and therefore; explain their presence in the forest.

The change in climate during the last 20 years has produced conditions that favor *P. leucopus* and have led to increases in their populations. Interestingly, *P. leucopus* has been reported as the most abundant small mammal in the E. S. George Reserve for 8 years (Journal of Field Ecology 1998-2000, 2006-2009). Population abundances appear to fluctuate annually. However, there are years where *P. leucopus* populations are extremely high and followed by years with exceedingly low abundances. This pattern seems to repeat approximately every 2 or 3 years. However, data from all of the 8 years in which *P. leucopus* was sampled are needed to determine a representative oscillating trend for its abundance.

It has also been suggested that populations of *P. leucopus* increases following short winters and decreases following prolonged winter periods (Myers et al. 2005). If this assertion is correct, then the year 2005 must have had a shortened winter period because *P. leucopus* was abundant the following year. The year 2006, however, may have had a prolonged winter period because *P. leucopus* populations were exceedingly low the following year. To confirm these assertions, weather and temperature conditions from the past sampling years should be collected and correlated with *P. leucopus* abundance. Any future small mammal surveys should also collect weather and temperature conditions as part of a larger experiment to pin-point an actual abundance model for *P. leucopus* over the years.

ACKNOWLEDGEMENTS

I thank Judith Wan and Phil Myers for the materials and preparation needed to complete this project, as well as Ana Chara for her clarifications on a previous study and supportive comments on the statistical analysis of this manuscript. I would also like to acknowledge Dave Allen for editions to this manuscript as well as John Vandermeer and Ivette Perfecto for their supportive feedback during the progress of this project.

LITERATURE CITED


OWL SURVEYS IN THE E.S. GEORGE RESERVE

RACHEL CABLE

ABSTRACT

A commonly overlooked ecological aspect of the E.S. George Reserve is the nighttime, and along with it, the creatures of the night. A census of the owls of the reserve and where they are located had yet to be performed. There are three common owls that permanently occupy the reserve: the eastern screech-owl (*Megascopes asio*), the great horned owl (*Bubo virginiansus*), and the barred owl (*Strix varia*), which I planned to census using the habitat-based fixed-radius point count method and broadcast call method. Three individuals, all eastern screech-owls, were either heard or sighted on two nights in two habitats within close geographic proximity to each other. Presence of owls in this area of the Big Woods may be due to the known preference of owls for old forests over younger forests. Future studies should take into consideration the breeding season of the target owl species, weather conditions, and moon phase as well as focus on old deciduous forest within the reserve to guarantee more call back success.

INTRODUCTION

Although the E.S. George Reserve has been studied from almost all ecological perspectives, one area largely remains a mystery: the night. Surveys and experiments at night can provide information about systems that are rarely encountered during the day, including nocturnal mammals, nocturnal plants and their pollinators, and nocturnal birds, more specifically, owls. As birds of prey, owls are top predators of their food chains, and thus their presence or absence, and changes in abundances, within their preferred environments can indicate the health of those environments (Grosshuesch 2006). The owl species that inhabit the ESGR and their seasonality are known (Payne 1997), but what habitats in the reserve they occupy and in what abundances had yet to be surveyed.

A simple method for surveying the presence and abundance of birds in a given area is by performing fixed-radius point counts. For this method, the observer records the number of individuals of each species heard calling within a certain radius of a chosen site within a certain time. Calls heard outside the radius and walking between sites can also be noted. When non-overlapping observation sites are evenly spaced throughout the area, abundances of species and their locations can be calculated from the resulting census data (Hutto et al. 1986).

Two types of fixed-radius point counts can also be performed based on where survey sites are chosen: population-based and habitat-based. Population-based point counts are performed on randomly chosen sites throughout a geographic area, and results are indicative of species within the geographic area, but not of the choice of habitats within the area. Habitat-based point counts are performed at sites that are specifically selected because of their habitat type, and thus results from these surveys can describe the species found within certain habitats (Huff et al. 2000). Another method for surveying for the presence of birds is to broadcast conspecific calls and
listen for callbacks, which has been proven effective for censusing many owl species (Takats and Holrod 1997, Zuberogoitia and Campos 1998).

The ESGR is known to house five species of owls, three of which are common permanent residents (Eastern Screech-owl, Great Horned Owl, Barred Owl) and two of which are uncommon or rare seasonal residents (Long-eared Owl, Northern Saw-whet Owl) (Payne 1997). The eastern screech-owl, *Megascopes asio*, is known to inhabit many environments, including woodlots, forests, swamps, parks, and suburban gardens, and is “best located and identified by [its] voice.” The great horned owl, *Bubo virginianus*, also inhabits many environments, ranging from forest to city to open desert, and nests in trees, caves, and on the ground. The barred owl, *Strix varia*, inhabits dense coniferous or mixed woods forests in river bottoms and swamps, and upland forests (Dunn and Alderfer 2006).

My survey utilized the habitat-based point count method as well as the broadcast call method in attempt to determine the habitats and abundances of the three most common species on the ESGR. The survey occurred over four types of habitats within the reserve: pine plantation, oak-hickory forest, old fields and edges, and swamp forest. From the described habitats, I expected to find the eastern screech-owls and great horned owls in all sites and the barred owls in the swamp forest sites.

**METHODS**

Seven sites, two representing pine plantation and swamp forest each, one oak-hickory, one old field and edges, and one both oak-hickory forest and old fields and edges, were chosen from within the reserve. Each was surveyed once over the nights of 15 October and 16 October 2010 from 1:00 to 3:00 on the first night and 22:30 to 3:00 on the second night (Fig. 1). The first night was in the 1st quarter moon phase and the second was in the waxing gibbous moon phase. At each site, a listening point was chosen where the observer could estimate a radius for birdcalls. In the pine plantations, the estimated radius was to the edge of the plantation.

Each survey consisted of four stages; 1) the observer listened in silence for ten minutes, 2) played one species’ call three times, then listened for a response for one minute, 3) played the next species’ call three times, then listened for a one minute, 4) played the third species’ call three times, finally listening for five minutes for any responses. If there was a response to a certain species’ call, it was broadcast again, with pauses for listening for responses, until the individual approached close enough to sight or stopped calling. This procedure was done once at each survey site. Any other wildlife heard or seen was also noted.
RESULTS

After surveying each site once, only one species of owl was heard and sighted on the reserve: the eastern screech-owl. The three individuals heard began calling after the broadcast of conspecific calls, and one of these later approached while calling. This occurred in two sites: the eastern pine plantation site on the first night and the eastern oak-hickory forest site (within the Big Woods) on the second night.

Evidence of other nocturnal wildlife was also observed. Coyotes were heard howling and fighting across the reserve. One white-footed mouse was sighted near the northern swamp site, and one cotton-tail rabbit was sighted in the western pine plantation. Many other unidentified calls were also heard throughout the reserve.

DISCUSSION

Although a correlation cannot be made from my study between the presence of certain owl species and certain habitats, a pattern of the geographic location of the three eastern screech-owls observed each night can be inferred. The pine plantation where the screech owls were first heard is geographically east of the Big Woods, the oak-hickory forest where the screech owls were observed the second night. It is likely that these owls reside in the Big Woods, as it is an older
stand of woods within the ESGR, and it has been shown that other species of owls prefer old forests (> 80 years old) to younger forests (such as the western oak-hickory forest surveyed, which was recently a field) (McGarigle and Fraser 1984, Mazur et al. 1998). A similar study performed using radiotelemetry to track eastern screech-owls in Kentucky showed that they vary their habitat use by season and tend to use deciduous woodlots during the non-breeding season (around May to January). This was postulated to be because these woodlots are an abundant source of prey and perches for the screech owls’ “sit-and-wait” approach to hunting (Sparks et al. 1994).

Many factors could have attributed to the lack of any other owl calls or call backs during the survey, despite the witnessed activity of owl prey (white-footed mice and cotton-tail rabbit) and evidence that both barred and great horned owls are known to respond to broadcast calls (McGarigal and Fraser 1984, Mosher et al. 1990, Morrell and Yahner 1995). A study done by Takats and Holroyd (1997) showed that owl response to broadcast calls could be affected by the temperature, with more responses at higher temperatures, and at least one of the survey nights had close to freezing temperatures (33º F in Howell, MI), which may have lead to low owl activity. Takats and Holroyd also reported that response to calls was much higher just after dusk (20:00 – 23:59) and just before dawn (4:00 – 7:59), and all of my surveys were made between 22:30 and 3:00, missing these optimal times. However, another study broadcasting to only great horned owls had the most responses after midnight and attributed the amount of responses more to the phase of the moon (best when waxing) and the clarity of the night (clear nights are better) (Morrell et al. 1991).

The strongest factor in response success to broadcast calls seems to be the season in which the survey is performed. Most studies recommend using broadcast calls just before or during breeding season, which lasts from February to April for eastern screech-owls (VanCamp and Henry 1975), late December to mid-February for great horned owls (Baumgartner 1938), and from April to August for barred owls (Mazur et al. 1998). Baumgartner also reports that great horned owl females are very rarely heard outside of the breeding season. The responses heard from eastern screech owls on the reserve outside of their breeding season could be explained by the dispersal of fledgling males, which begins in May and can end in October, where males call to protect territories and establish new ones (Belthoff and Ritchison 1989).

Another potential problem in my survey was the fact that I did not know anything about the conspecific calls I was broadcasting: whether they were made by a male or female, whether they were territorial or mating calls, whether they were made by mature or immature individuals. Eastern screech-owls use two calls, the bounce and the whinny, in different combinations to call for mates (bounce and whinny) and to show aggression (perhaps whinny only) (Ritchison et al. 1988). The call I was using for the eastern screech owl may have been a male mating call, repelling males and attracting females, or a male aggression call, attracting the male whose territory I was broadcasting in and repelling other males. Klatt and Ritchison (1993) also describe a complicated duetting system for finding and identifying mates between eastern screech owl males and females that might be hard to replicate via recordings. The male great horn owl is also known to have many different calls that vary based on the season and territoriality, which include non-breeding season “chatter” (Baumgartner 1938), a call which
may have been heard on the reserve the weekend before my study. Any of these calls could have been the call I broadcast.

For future owl surveys on the reserve, the seasonality of the species being targeted, temperature, moon phase, and timing during the day should be taken into account. According to other surveys of great horned and barred owls, they both prefer old or mature deciduous forests as habitats (Nicholls and Warner 1972, Morrell and Yahner 1994), indicating that another survey should focus more intensely on the areas within the reserve that match this description. Also, other methods such as searching prospective nesting sites for owl nests may aid in surveying owls that are more easily flushed and do not reply to broadcast calls (Zuberogoitia and Campos 1998). These censuses should be performed over many breeding seasons to determine the true abundances and locations of the owls of the preserve.

ACKNOWLEDGEMENTS

I would like to express my gratitude to David Gonthier for performing the role of the lovely assistant with grace and glamour.

LITERATURE CITED


EFFECT OF BARBERRY ON LEAF LITTER ASSOCIATED INVERTEBRATES

ANA CHARÁ

ABSTRACT

This study was performed in order to assess the differences of leaf litter characteristics and invertebrate community composition in sites under barberry clumps and in sites with no barberry within the Big Woods in the E.S. George reserve. Sampling was performed in four 1 m² quadrants in each habitat type, where environmental parameters like soil pH and humidity were measured, the leaf litter was completely removed and the invertebrates were sorted. There were no differences in the composition and diversity of the invertebrate community associated to leaf litter under barberry compared that of the rest of the forest which can be attributed to the lack of differences in microhabitat conditions such as pH, moisture and leaf litter diversity. Due to the low sampling size no robust conclusions can be addressed and further research must be carried out in order to determine the accuracy of these results.

INTRODUCTION

Litter production and decomposition are fundamental ecological processes because they play key roles in nutrient cycling and turnover of carbon and nutrients in terrestrial ecosystems (Tang et al. 2010). Forest floor conditions such as humidity, pH and physical and chemical composition of the litter determine the composition of the decomposer community, and in turn, this community affects the balance between mineralization and immobilization of nutrients in litter and leachates (Burghouts et al. 1992).

The japanese barberry (Berberis thunbergii) is a perennial shrub native to central and southern Japan that has been reported as an invasive species in relatively undisturbed forests of the northeastern United States (Cassidy et al. 2004). It forms dense monospecific populations that can affect ecosystem function in several ways, such as the displacement of native plants for shading, and reduction of wildlife habitat and forage (Swearingen 2009).

Several studies have proved that the dense clumps of barberry also cause negative effects in soil characteristics of the forests, such as the increase of the pH level, the reduction of the depth of the litter layer and the alteration of the microbial decomposer community (Kourtev et al. 2002). But no studies have been performed to assess the effect of this invasive species in the leaf litter invertebrate community, which must respond to the changes it causes in the microhabitat quality.

This study was performed in order to assess the differences of leaf litter characteristics and invertebrate community composition in sites under barberry clumps and in sites with no barberry within the forest. I expected to find that volume of leaf litter, leaf diversity, pH and moisture will
differ under barberry compared to the forest and that in consequence the invertebrate community will be different.

METHODS

The study was conducted in a section of the Big Woods of the E.S. George Reserve in Pinckney, Michigan, adjacent to Esker Road. The Big Woods is a forest dominated by oaks and hickories, with an understory consisting primarily of *Prunus serotina*, *Acer rubrum*, and large clumps of barberry.

Data was collected on October 17 of 2010, after approximately 60% of the autumn leaf fall. Sampling was performed placing four 1 m² quadrants under dense barberry clumps and four 1 m² quadrants in other places in the forest without barberry. In each quadrant environmental parameters like soil pH and humidity were measured in three random points and averaged, and the leaf litter was completely removed and transported to the laboratory.

In the laboratory, invertebrates were sorted, leaf litter was weighed and the number of leaf species in each sample was recorded. Invertebrates were preserved in 70% ethanol, identified at order level and sorted to morphospecies.

Invertebrate richness and density (number of individuals/g leaf litter), soil moisture and pH, and leaf litter richness and biomass, were compared between the samples under barberry (here forward referred as barberry) and the samples in the rest of the forest using t-student analysis. Regression analysis was used to test the relation between habitat characteristics (such as soil moisture and leaf species richness) and the abundance of leaf litter invertebrates. All analysis were performed using PAST and SPSS.

RESULTS

Invertebrate community

A total of 40 individuals of 8 orders and 14 different morphospecies and were collected. The orders sampled included mites (Acarida), two orders of springtails (Entomobryomorpha, Symphypleona), spiders (Araneae), beetles (Coleoptera), snails (Gasteropoda), ants (Hymenoptera) and moths (Lepidoptera).

More individuals and more species were collected in the forest than in the barberry but the differences were not significant (Table 1). However, there were more individuals per gram of leaf litter in the barberry than in the forest, but again the difference was not significant (Table 1).

<table>
<thead>
<tr>
<th>Measure</th>
<th>Barberry</th>
<th>Forest</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total Individuals</td>
<td>19</td>
<td>21</td>
<td>0.98</td>
</tr>
<tr>
<td>Species richness</td>
<td>11</td>
<td>12</td>
<td>0.72</td>
</tr>
<tr>
<td>Mean Individuals/g leaf litter</td>
<td>0.06</td>
<td>0.04</td>
<td>0.39</td>
</tr>
</tbody>
</table>
The composition of the invertebrate community under barberry was similar to that found in the rest of the forest (Figure 1). Only three morphospecies were found in the forest and not under the barberry; and two morphospecies were found in the barberry but not in the forest. The family Entomobryidae (Springtail) was more abundant in the forest than under barberry, and the ants of the genus *Prenolipes* were more abundant in the barberry than in the forest, but there were no significant differences in any of the two cases.

![Figure 1. Composition of invertebrate community in the studied habitats](image)

Leaf litter characteristics
Both sampled habitats presented an average of 5.7 leaf species per quadrant. Average leaf litter biomass was higher in the forest (209.38 g) than under barberry (120.19 g), but there were no significant differences ($p = 0.1$), probably due to the big variation of the biomass in the forest (Figure 2).
Soil moisture was very similar between the barberry samples and the rest of the forest ($p = 0.5$) (Figure 3). There was no variation in pH values among the samples.
Relation between invertebrate abundance and habitat characteristics

There was no relation between leaf litter species richness and invertebrate abundance ($p = 0.6$). However, the abundance of invertebrates showed a tendency to increase with moisture (Figure 4), but the regression value was not significant ($p = 0.1$).

![Figure 4. Variation of invertebrate abundance as a function of soil moisture](image)

DISCUSSION

I did not observe significant differences in the composition and diversity of the invertebrate community associated to leaf litter under barberry compared that of the rest of the forest. This can be attributed to the fact that the conditions of the microhabitat such as pH, moisture and leaf litter diversity did not present significant variations between the sites with barberry and the sites with no barberry, and therefore did not cause any shift in the composition of the associated invertebrates.

However, there were some interesting tendencies, like the lower biomass of leaf litter under barberry, which supports results obtained in previous studies (Kourtev et al. 2002) and can be explained by the tick canopy cover of the barberry which may act like a trap, preventing leaf litter to reach the soil. The lack of significant differences for this parameter is likely due to the low sample size which causes high variation in the results.

Even though the results indicate that the barberry is not affecting the leaf litter processes in the forest, the sample size was too small to make robust conclusions. A previous study of leaf litter arthropods in the E.S. George reserve found 54 individuals of 30 morphospecies in the mixed forest, which compared to the 14 morphospecies found in this research indicate that I did not obtain a representative sample of the community. Another important factor is the time of the year when the study was performed, after more than 50% of the leaves had fallen and the temperature was beginning to drop, which is likely to cause changes in the activity of litter invertebrates and composition of the leaf litter. Furthermore, the instruments used to measure pH and moisture, were not the proper ones to make this type of measurements and might be introducing error on the data.
However, the lack of differences in the leaf litter invertebrate community has been reported previously in studies performed in the reserve, Li et al. (2008) found that the arthropod community did not differ between an oak forest, and old red pine plantation and mixed forest. They attributed this result to the high connectivity of the arthropod community in the George reserve and this may be also an explanation for the lack of differences in the invertebrate community found in this study.

ACKNOWLEDGEMENTS

Special thanks to David Allen for preparing all the materials and tools needed for this study and to John Guittar and William Webb for the company during the sampling.

LITERATURE CITED


THE EFFECT OF DISTURBANCE RECOVERY TIME ON FOREST LEAF LITTER

SUSAN CHENG

Abstract: Leaf litter is an essential part of an ecosystem because its decomposition continues the cycling of nutrients in that habitat. Natural and human-caused disturbances can change the characteristics of leaf litter in forests. To determine the impact that recovery time has on disturbed sites, I measured the diversity and biomass of leaf litter in two different aged forests. Results showed that leaf litter diversity was higher in the older forest, although this was not statistically significant. In addition, there was more leaf litter biomass in the younger forest than there was in the older forest (p<0.01). The higher amount of leaf litter in the more recently disturbed site may be a result of a higher decomposition rate in the older forest or the dominance of more difficult to degrade Quercus spp. leaves in the younger forest.

INTRODUCTION

Leaf litter influences ecosystem functioning in many ways. For instance, the decomposition of leaf litter converts stored carbon, nitrogen, and other nutrients in leaves into available forms for other organisms to use (Hättenschwiler 2005). This in turn determines which organisms can exist in a habitat and how productive that ecosystem’s vegetation can be (Jonsson and Wardle 2008). Decomposition of leaf litter has been observed to increase with leaf litter diversity (Hättenschwiler 2005). In addition, the physical properties of leaf litter affect community structure. The shapes and sizes of leaves define the temperature and hydrological characteristics of the forest floor and thus, the types of organisms that can inhabit that leaf litter (Hättenschwiler 2005). Previous studies have also shown that the diversity of leaf litter influences the characteristics of that litter. A heterogeneous leaf litter will have different chemical and physical properties compared to one comprised of one species of leaves, which affects decomposition rates and nutrient availability (Gartner and Cardon 2004).

Disturbance is another variable that influences ecosystem productivity and structure (Xi et al. 2008). The stage of succession a forest is in defines the type of plant community, decomposer substrates, and leaf litter that is present in that forest (Fisk et al. 2002). Previous research has shown that logged riparian areas have less leaf litter mass loss and that old-growth forests have greater litter mass loss and nitrogen accumulation (Mayer 2008, Kreutzweiser 2008). On the other hand, litter amounts could also be lower in disturbed areas (Xu et al. 2004).

Since leaf litter strongly affects ecosystems, this study was developed to understand how leaf litter characteristics change as a forest recovers from a disturbance. The litter characteristics I chose to examine were species diversity and leaf litter weight as a proxy for biomass. Since older forests have more time to accumulate species of trees and thus more leaf litter, I predicted that older forests would have more leaf litter diversity and biomass as compared to younger forests.
METHODS

This study took place at the E.S. George Reserve in Livingston County, MI in October 2010. Two forested areas that underwent disturbances at different times were chosen as study locations. The first site was in the Big Woods and represented a forest with a longer recovery time. It is an oak-hickory dominated forest that experienced cycles of burning until approximately 150 years ago. In contrast, the West Woods has experienced more recent disturbances. After 1927, the West Woods was no longer used as pasture or farmland and has since then had time to undergo succession. In addition, the portion of the West Woods that was used in this study was in the proximity of a tornado that is estimated to have gone through the area in 1997.

At each forest site, three 50-meter (m) transects were laid parallel to each other. A 0.5 x 0.5 m quadrat was placed next to the transect tape at 0, 20, and 40 m. At each quadrat, I identified the number of species represented in the leaf litter. The leaf litter was then bagged and dried in a Grieve oven for approximately three days. I then weighed the contents of the bags using a Mettler-Toledo PB3002-S scale.

Estimate S and Excel were used to calculate the species accumulation curves and to run statistical analyses on the diversity and biomass data.

RESULTS

Species Diversity

The Big Woods had two more species of trees than the West Woods did (Table 1). Six of the tree species found in the West Woods were also found in the Big Woods. These included *Quercus rubra/velutina*, *Sassafras albidum*, *Acer rubrum*, *Carya spp.*, *Prunus spp.*, and *Ulmus americana*. In addition to these species, *Elaeagnus umbellata* was present in the West Woods leaf litter. *Quercus alba*, *Cornus florida*, and *Hamamelis virginiana* were found in the Big Woods leaf litter but not in the West Woods.

Table 1: Tree species found in the Big Woods and West Woods leaf litter.

<table>
<thead>
<tr>
<th>Big Woods</th>
<th>West Woods</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Quercus rubra/velutina</em></td>
<td><em>Quercus rubra/velutina</em></td>
</tr>
<tr>
<td><em>Sassafras albidum</em></td>
<td><em>Sassafras albidum</em></td>
</tr>
<tr>
<td><em>Acer rubrum</em></td>
<td><em>Acer rubrum</em></td>
</tr>
<tr>
<td><em>Carya spp.</em></td>
<td><em>Carya spp.</em></td>
</tr>
<tr>
<td><em>Prunus spp.</em></td>
<td><em>Prunus spp.</em></td>
</tr>
<tr>
<td><em>Ulmus americana</em></td>
<td><em>Ulmus americana</em></td>
</tr>
<tr>
<td><em>Quercus alba</em></td>
<td><em>Elaeagnus umbellata</em></td>
</tr>
<tr>
<td><em>Cornus florida</em></td>
<td></td>
</tr>
<tr>
<td><em>Hamamelis virginiana</em></td>
<td></td>
</tr>
</tbody>
</table>

Although the Big Woods had a greater number of species of trees than the West Woods did, this difference was not statistically significant (Figure 1).
The West Woods had significantly more leaf litter than the Big Woods did (p<0.01, Figure 1). The average dry weight of leaf litter found in a 0.25 m² area in the Big Woods was 65.79 g while the average dry weight of leaf litter in the West Woods was 96.39 g.
DISCUSSION

Results showed that the diversity between the two different aged forests recovering from a disturbance was not significantly different. However, I observed that there was a more even distribution of leaves per species in the Big Woods and that the vast majority of the leaves in the West Woods were comprised of *Quercus rubra/velutina*. Thus, the addition of abundance of leaves per species in future studies would improve this measure of diversity.

Analysis also showed that the more recently disturbed West Woods had a greater dry leaf litter weight than the older Big Woods. Although this was contrary to my expectation, this supports observations that old-growth forests have higher litter mass loss than transition forests or old fields (Mayer 2008). In addition, this trend can be explained by the conditions that control decomposition in an ecosystem. The rate of decomposition is mediated by climate, litter quality, and the environment, with the last factor strongly influencing the decomposer community (Jonsson and Wardle 2008). Since the Big Woods is older, it has had more time to develop a diversity of substrates and nutrients for decomposers. With more decomposers in the Big Woods, leaf litter can decay much faster. In fact, Sylvain et al. 2010 found a greater fungi diversity and abundance in deciduous forests as compared to an old field. An alternate explanation for the higher dry weight in the West Woods is that *Quercus spp.* leaves contain compounds that are harder for microbes to degrade (Steffen et al. 2007).

Future research should include analysis on the abundance of each leaf species as well as decomposer abundance and diversity. Together, this cohesive collection of ecosystem data can improve our understanding of leaf litter dynamics in aging forest ecosystems.

ACKNOWLEDGEMENTS

Many thanks to Jim Le Moine, Dave Allen, John Vandermeer, and Ivette Perfecto for resources and intellectual support.
LITERATURE CITED


Mayer, P.M. 2008. Ecosystem and decomposer effects on litter dynamics along an old field to old-growth forest successional gradient. Acta Oecologica 33: 222-230.


OBTAINING AND APPLYING SIMPLE ESTIMATES OF SAPLING BIOMASS

RAFAEL D’ANDREA

ABSTRACT

Tree biomass is an important quantity which is obviously difficult to measure directly. Notwithstanding the rich literature on the relation between plant biomass and a host of other, more easily obtainable variables, we pursue the simplest way of achieving reliable estimates. Using data from red oak saplings collected in the ES George Reserve, we achieve a simple, elegant formula with a very high goodness-of-fit, which stands the test against independent datasets from previous work. Should it be confirmed by further research, our result is likely to become a useful and readily applicable tool on studies concerned with sapling biomass.

INTRODUCTION

Estimates of biomass in forests are useful and important for many purposes, including but not limited to forest management and tracking carbon stocks (Brown & Lugo 1992). As direct measurements of biomass are destructive and often times impractical, the most common strategy is to seek indirect means. Several methods can be applied (Brown, Gillepsie & Lugo 1989), and one of the simplest is to establish the relation between biomass and non-destructive measurements, diameter being probably the easiest to obtain. Previous studies of differing levels of sophistication have explored this line of inquiry, to differing results (Araujo et al 1999, Ketterings et al 2001). Here we are interested in the simplest possible model of such kind, namely, \( W = \alpha D^k \), where \( W \) is above-ground fresh biomass, \( D \) is some measure of its diameter, and \( \alpha \) is a parameter related to density. Whereas the majority of the literature is concerned with adult trees, we base our investigation on saplings.

MATERIALS AND METHODS

Each sample had its height measured in situ with a measuring tape and, after being cut and removed, had its basal diameter measured with a caliper. A few hours after harvest, the entire plant was weighed. Leaves were not dried prior to weighing.

Two plots were used for harvest, one in Evan’s Old Field and the other in the Big Woods. A total of thirty saplings were collected, fifteen from each plot. The samples, all from species in the red oak group, fell in a height range of 18.5 to 101.0 cm, and a basal diameter range of 3.4 to 27.2 mm.
We test correlations between weight\(^1\) (W) and a range of power functions of height (H) and basal diameter (D) in order to establish the best match. We find the best fitting exponent \(k_{\text{best}}\) to the model \(W \sim D^k\). We examine how H affects the correlation for the case \(W \sim D^2 H^m\).

**RESULTS**

Fig 1 shows that H is clearly linear on D, as expected from a model where saplings of different sizes are regarded as rescaled versions of a template shape. Assuming such fixed shape, one expects \(k_{\text{best}} = 3\) (this can be easily seen for the simplest possible case, a circular cylinder: the volume is proportional to the cross section area, \(\pi D^2 / 4\), multiplied by the height, H. As the latter is linear on D, one gets a cubic relation, \(W \propto D^3\)). This is not, however, what is found. Instead, \(k_{\text{best}}\) is much closer to 2, as can be seen from Fig 2 and from comparing Figs 3, 4 and 5.

A side effect of such a good fit of weight on the square of the diameter is that including any power of the height H yields a worse fit (Fig 6). Therefore, our results dispense with height altogether, and our phenomenological result is then

\[
W = \alpha D^2.
\]

---

\(^1\)For operational simplicity, we include moisture in our measure of biomass.
Figure 2. Goodness of fit (measured as Pearson’s r) of the model $W \sim D^k$ for a range of values of k. The strongest correlation occurs just below $k = 2$.

![Weight ~ (Diameter)^k](image)

Figure 3

![Weight X Diameter](image)

Figure 4

![Weight X Diameter^2](image)
DISCUSSION

Our formula, though seemingly very accurate, departs from expectations of the “template shape” simple model. Assuming constant density throughout the plant’s tissue, the possible causes for such deviation must lie in changes in shape as \( D \) varies. One of the most obvious such deviations is uneven tapering. As long as one assumes the rescaled fixed shape model, tapering does not change the mathematical relation \( W \sim D^3 \). It can be shown, however, that if tapering correlates positively with \( D \) (i.e., if the ratio (diameter at tip)/(basal diameter) decreases as basal diameter increases), then indeed \( k < 3 \). The reverse is true if tapering correlates negatively with \( D \). The effects of differential tapering on the exponent \( k \) are not, however, strong enough to explain our experimental result, \( k_{best} \lesssim 2 \).

A more likely reason is branching. The total biomass of the tree is obviously the sum of the biomass in its trunk and in the branches and leaves. As a tree grows, both basal diameter and the
number of branches (and leaves) increase. Therefore, the tree shape clearly changes as \( D \) increases (much more radically than any alteration caused by uneven tapering). The branching pattern and the biomass input rate on each branch would have to conspire to a very particular combination in order to save the cubic scaling; most likely, once branching enters the model biomass ceases to even scale as a power law on \( D \). Curiously, our data points fit very well to a power law (Fig4), an account of which is left for future research to provide.

Setting apart the underlying reasons for the pattern, we can apply it to support the conclusions of a previous work (D’Andrea et al 2010a), which assumed, without justification, that biomass input rate in red oak saplings scales with the square of basal diameter. The results obtained here provide validation to their conclusion that biomass allocation in these young trees varies according to the environment in which they grow.

As our study was based on data obtained exclusively from oaks, it does not in principle purport to be extrapolated to unrelated genera of trees. Nevertheless, a case could be made for such generalization, since the relation \( W = aD^k \) hinges essentially on general morphology, which is arguably similar across most taxa.

Another example where our empirical relation finds utility is D’Andrea et al 2010b. In that study, leaf litter mass contained in quadrats on the floor was shown to correlate with, among other predictors, the total diameter at breast height (DBH) of trees in a circle surrounding the quadrat. That leaf litter and DBH of surrounding trees should correlate positively is clear, as the litter stock correlates with the amount of leaves in the canopy, which in turn correlates with DBH. The harder question is to which power of DBH litter should correlate linearly with. The authors anticipated a cubic scaling, but were puzzled that the correlation proved stronger with \((DBH)^3\) than with \((DBH)^3\).

It seems reasonable to suppose that litter stocks scale linearly with the mass of leaves in the canopy. The question then becomes how leaf production correlates with DBH. At first it could be assumed, as done in that paper, that it scales with biomass. If that is the case, then our relationship is

\[
\text{litter} \propto \text{biomass} \propto DBH^k
\]

A more biologically informed hypothesis is that it scales with metabolism, as most of the biomass in a tree is dead matter, which does not produce leaves. According to this hypothesis, the power we are looking for is given by

\[
\text{litter} \propto \text{metabolism} \propto \text{biomass}^b \propto DBH^{kb}
\]

where \( b \) is the allometric exponent for metabolic scaling. There is no shortage of literature on the matter of metabolic allometry in living organisms in general and vascular plants in particular, and though there is no agreement on the value of \( b \) or even that such a universal number exists (Agutter & Wheatley 2004, Reich et al 2006), the majority of theoretical (West, Brown & Enquist 1997, 1999) and experimental (Reich et al 2006, Cheng et al 2010, Mori et al 2010) studies advocates some value in the [3/4, 1] ballpark. A significant minority of investigators (Heusner 1982, Harvey & Bennet 1983) holds even that \( b \) is actually closer to 2/3, though their case seems to be more applicable to animals than plants.
If biomass is naively assumed to scale with \((DBH)^3\), then the best exponent would be either equal to 3 (according to the leaves \(\propto\) biomass hypothesis) or fall within the range [2.25, 3] (according to the leaves \(\propto\) metabolism hypothesis). Nevertheless, the data supported a linear correlation much more strongly than either a cubic or a square correlation, which poses somewhat of a conundrum. The mystery dissipates as one applies our result, \(k = 2\). In that case, the expected power drops to 2 (leaves \(\propto\) biomass hypothesis) or [1.5, 2] (leaves \(\propto\) metabolism), or even to 1.3, if the true allometric exponent for these trees is 2/3. Fig 7 displays Pearson’s r between the leaf litter and a range of powers of DBH. The plot leaves little doubt that application of our result is an improvement over both the naive assumption \(k = 3\) and the biologically and mathematically unfounded assumption \(k = 1\) used in D’Andrea et al 2010b.

Figure 7. Correlations between leaf litter and DBH from a group of adult trees commonly found in the ES George Reserve, of genera outside of Quercus. Data from D’Andrea et al 2010b.

CONCLUSIONS

Despite the existence of formulas in the literature relating biomass to other quantities in woody plants, our study provides an expression which, in spite of its extreme simplicity, seems to have a very strong hold on our data. Other works have pursued the same strategy and found somewhat different results (for instance, Ketterings et al (2001) found several values for different groups of adult trees in secondary forests in Sumatra, with \(k\) ranging from 2.3 to 2.6). A possible explanation for the discrepancies are that our result was obtained exclusively from saplings. This is in contrast, however, with our hypotheses that \(k\) should depart further from 3 with increasing branching.

Should further studies confirm and generalize our phenomenological power law obtained for oak saplings, we will have a crude but efficient tool to be used when woody plant biomass needs to be estimated. All it requires is the diameter of the main stem and calibration to determine the one free parameter. The fact that the correlation seems to be very strong \((R^2 = 0.95)\) adds to its
appeal and calls for theoretical justification, which we suggest should be based on branching patterns in trees.

LITERATURE CITED


S. Mori et al. Mixed-power scaling of whole-plant respiration from seedlings to giant trees. PNAS 107, 1447-1451 (2010).


POND CONNECTIVITY AND THE ABUNDANCE OF ADULT SALAMANDERS IN THE E. S. GEORGE RESERVE

DAVID GONTHIER

ABSTRACT

Metapopulation dynamics contribute to population size at a local scale. Therefore each sub-population must be put into a broader context, where the size of adjacent populations might influence the given sub-populations through dispersal. Amphibian larval populations in aquatic ponds are model systems to study the importance this connectivity between populations. In this study, I investigated whether the connectivity and larval population size of ponds within eight different 9-hectare sites could explain the abundance of adult salamanders occurring within the terrestrial component of those sites. Adult salamander populations were estimated with three different variables: occupation rate of logs per site, the number of salamanders found per site (21 log survey), and the number of *Ambystoma laterale* per site (21 log survey). Data from a larger project was used to estimate salamander connectivity between ponds, as well as, local larval population sizes within each site. Connectivity and larval population size only explained marginal amounts of variation in occupation rates. However, site connectivity explained a significant amount of variation in salamander density per 21 logs, yet larval density did not. Interestingly, when only *A. laterale* was considered larval population size explained significant (in some cases marginal) amounts of variation in density of adult salamanders and connectivity did not. The three measures of density appeared to respond differently to connectivity and larval density. The results of this study suggest that the connectivity of sub-populations, as well as, neighboring larval populations are factors that can contribute to the patchiness of adult salamander populations across habitats.

INTRODUCTION

Ecologists have long sought to understand the factors that explain the abundance and diversity of organisms across ecosystems. Metapopulation dynamics are important to the local dynamics of organism through regional processes of migration, colonization, and extinctions within and between local subpopulations (Levin 1974, 1976, Leibold et al. 2004). For a given metapopulation the success of subpopulations may therefore depend on its connectivity to other subpopulations (Werner et al. 2009, Marsh & Trenham 2001). Therefore, population density of adjacent subpopulations, dispersal ability of a given species, and the distance between subpopulations are important variables that can contribute to our understanding of local population dynamics.

Amphibian communities are particularly good systems to study metapopulation dynamic processes because larvae are dependent on aquatic habitats making ponds essentially ‘islands’ that adult amphibians must disperse between (Werner et al. 2009, Marsh & Trenham 2001, Smith & Green 2005). Additionally, local extinction and reestablishment of ponds are important dynamics of amphibian populations. Droughts may cause ponds to dry up entirely causing local extinctions of amphibians, but after ponds is rehydrated by years of normal rainfall, re-
colonization of that pond by dispersing adults can reestablish local populations (Werner et al. 2009)

Previous studies demonstrate that estimations of connectivity that account for distance between ponds, dispersal distances of adults, and larval populations sizes between anuran ponds can explain significant amounts of variation in both larval and adult Anuran populations (Werner et al. 2009). However, the importance of patch connectivity to salamanders, have not yet been considered. The purpose of this study was to determine if regional connectivity processes could explain the abundance of adult salamanders within a landscape. I predicted that like anurans, adult salamander density would correlate with connectivity.

METHODS

Adult salamander survey

This study was conducted at the E. S. George Reserve in October of 2010. First, I studied the reserve map to find sites that varied in pond density. For each site location, I took a GPS unit and determined were on the map I was located. I then turned over fallen logs within the immediate area and counted the number of each salamander species present. A log was defined as any woody debris within a size class greater than approximately 15 centimeters in diameter and smaller than any log that could not be moved by the field team. I surveyed 21 logs per site for salamanders. Altogether I surveyed eight sites within the reserve. From these counts I calculated the proportion of logs occupied by salamanders, the total number of salamanders per 21 logs, as well as, the total number of each species of salamander found per 21 logs per site.

Figure 1. The E. S. George Reserve with each site location marked by its 9-hectare area.
Connectivity

To calculate connectivity for each site, I modified calculations from Werner et al. (2009). I obtained larval abundance data for most ponds within the reserve from a larger ongoing project at the E. S. George Reserve responsible for generating the data in Werner et al. (2009). Unfortunately, only data for *Ambystoma laterale* (blue-spotted salamander) were available, so I assumed that total larval salamander abundance correlated with blue spotted salamander abundance and therefore used *A. laterale* population size for calculating connectivity. A connectivity index ($S_i$) was calculated for all ponds within each site:

$$S_i = \sum \exp(-\alpha d_{ij})N_j$$

Where $S_i$ is the connectivity of pond i, $\alpha$ is equal to 1 over the mean dispersal distance of a given salamander species (100m, as reported for anurans in Werner et al. 2009), $d_{ij}$ is the distance between pond j and pond i, and $N_j$ is the larval population size of *A. laterale* within pond j for 2009 (Werner et al. 2009). The connectivity value S was calculated for all ponds within a 9 hectare square centered around each sampling site, which was estimated by overlaying a grid over the reserve map at the point where the 21 logs were surveyed at each site (Fig. 1). Note that
for this analysis only connectivity between ponds falling within a given site was considered, in other words, $d_{ij}$ was not calculated between pond $i$ and other ponds outside of the site area.

To estimate connectivity for each site a second connectivity index $C$ was created for each site:

$$C_i = \sum S_j$$

Where $C_i$, the connectivity per site $i$ (all ponds within 9-hectare area of the site) was then calculated as the sum of the connectivities of all ponds within the 9-hectare area of the site. Several problems occurred for certain sites because larval densities per area of a pond were not recorded or pond or swamp areas were not given. In these cases, larval density was assumed to be 0.25 larvae per m$^2$, which was a low density estimate for conservatism. Further in these cases pond area was estimated from a topographic map of the reserve (Fig. 1). Site 8 proved even more problematic because it was located on an island surround by a large swamp for which there was no area estimations and only one small bay had a population estimation. In this cause it was assumed that the large swamp was equal to four ponds the size of cattail marsh (larval density of 0.25 per m$^2$; area = 20560 m$^2$) the small bay located within the large swamp. Each hypothetical pond was located in each of corner of the 9-hectare square site and assumed to be 1m from one another (although they were obviously continuous through the swamp itself).

Larval population size

For each site, I also made an estimation of larval population size with the data set from Werner et al. (2009). The area of a given pond, swamp, or marsh was multiplied by the estimated density of $A. \text{laterale}$ larvae per m$^2$ of given body of water. I assumed that all salamander species correlated with population sizes of $A. \text{laterale}$. For each site, I summed the larval population size across all ponds that were present within 9-hectare area of the site.

Analysis

To compare site adult salamander density with estimates of connectivity and larval population size a number of comparisons were made. I estimated salamander density three different ways: log occupation by salamanders, total number salamanders recorded per site (21 logs), and total number of $A. \text{laterale}$ per site (21 logs). Of all species found, only $A. \text{laterale}$ salamanders were considered because it was relatively abundant and the estimations of connectivity were calculated from $A. \text{laterale}$ data. Finally, I determined how much variation was explained in these variables with site connectivity and larval population size using least square regressions. All statistical analyzes were conducted on SPSS (16.0).

RESULTS

In total 168 logs were surveyed, of which 14 were occupied by salamanders (8.3%). Eighteen salamander individuals were recorded from 4 species including 7 blue spotted ($A. \text{laterale}$), 1 spotted ($A. \text{maculatum}$), 1 Eastern red-backed ($Plethodon \text{cinereus}$), and 9 four-toed salamander ($Hemidactylium \text{scutatum}$). Sites varied in occupation of salamanders from 0 to 30 percent of logs occupied (Fig. 2).
Forty-four percent of the variation in log occupation was marginally explained by the connectivity within a given site (Fig 3a, $R^2 = 0.44$, df = 7, $F = 4.73$, $P = 0.073$). Whereas, 55 percent of the variation in log occupation was marginally explained by the larval population size within site (Fig 3b, $R^2 = 0.52$, df = 7, $F = 6.4$, $P = 0.052$). When the data was considered in a different way to include all salamanders found per site, 51 percent of the variation in salamanders per 21 logs was explained by site connectivity (Fig 4a, $R^2 = 0.51$, df = 7, $F = 6.2$, $P = 0.047$). Whereas larval population size marginally explained 41 percent of the variation in the number of salamanders per site (Fig 4b, $R^2 = 0.41$, df = 7, $F = 4.1$, $P = 0.09$). Considering all larval populations estimations and site connectivity values were estimated from data of only $A.$ laterale, I next considered adult $A.$ laterale populations alone. Site connectivity did not explain a significant amount of variation in $A.$ laterale per 21 logs (Fig. 5a, $R^2 = 0.08$, df = 7, $F = 0.5$, $P = 0.51$). However, larval population per site explained 51 percent of the variation in the number of $A.$ laterale per 21 logs (Fig. 5b, $R^2 = 0.51$, df = 7, $F = 6.2$, $P = 0.047$).

Figure 1. Percent log occupation by salamanders verses the connectivity (a) and larval population size (b) of sites.

Figure 2. Total number of salamanders per 21 logs verses the site connectivity (a) and larval population size (b).
DISCUSSION

These results provide marginal evidence that connectivity between ponds is an important factor explaining the abundance of adult salamanders in the terrestrial community. Pond connectivity had a stronger relationship than larval population size when total number of salamanders per site was considered, but not when log occupation or \textit{A. laterale} alone was considered. Both connectivity and larval population measures were included because it would be expected that if dispersal and distance between ponds were important to salamander populations, connectivity would produce better-fit regressions than larval population alone. Indeed, reviews suggest that pond isolation is an important variable that explains pond extinction in many systems (Marsh & Trenham 2001). Yet, my results are inconclusive, depending on the measure of adult salamander density, the two explanatory variables switch back and further in their ability to explain higher levels of variance. Nonetheless, like previous studies with anurans (Marsh & Trenham 2001, Werner et al. 2009), salamander adult populations correlated with measures of connectivity an important result to note.

There were many shortcomings to this study. Sample size was small within each site and larger surveys would undoubtedly provide better estimates and generate greater variance.
between sites. Additionally, the estimates of pond population size were flawed in many ways because of the limited data available for salamander populations that only included one species (*A. laterale*). The weak statistical relationships between log occupation and connectivity would likely be improved if all salamander population data were used for estimation of connectivity. It also should be noted that missing surveys from key ponds and swamps made estimations difficult and probably biased. Further, estimations of connectivity at the pond scale should include distances and population sizes of all ponds in the entire regional area (E. S. George reserve), not at the scale of site (9-hectares). However, given a mean dispersal distance of 100m, perhaps regional effects will be adequately captured at 9-hectares. Unfortunately, even this 100m distance is arbitrary because it was estimated from anurans. Yet, despite all these shortcomings, data on metapopulation processes are rare on scales such as these and these estimations still provide important contributions to the study amphibian populations.

This study also has conservation implications. If connectivity between patches are key functions of amphibian populations than fragmentation of habitats may be a factor contributing to the great decline currently being observed in amphibian populations (Marsh & Trenham 2001).

In conclusion, here I demonstrate that adult salamander populations within terrestrial communities are related to the connectivity of larval pond habitats. Further, this study provides another example of the importance of metapopulation dynamics to local populations.

**ACKNOWLEDGEMENTS**

I would like to give special thanks Rachel Cable for leading the salamander team survey, to Earl Werner, his lab, and collaborators for providing the data necessary to conduct connectivity analyses, and Ivette perfecto who is credited with the original project idea. John Vandermeer and David Allen provided important feedback and comments on project proposals and previous drafts. Finally, thanks to the 2010 Field Ecology class for their high level of awesomeness.

**LITERATURE CITED**


ABSTRACT

In the last five years, at least four studies have investigated the spatial distribution of the invasive shrub *Berberis thunbergii* in the E. S. George Reserve. Each study used different methods to answer the same fundamental question: how is *B. thunbergii* arranging spatially and why? In this study we review previous studies methods and results, ultimately corroborating previous findings that clump size vs. frequency roughly follows a power-law distribution. However, we acknowledge the fact that phenomena other than self-organization can lead to an apparent power-law distribution. Thus, we also propose a simplistic theoretical model for exponential colonization and growth of an invasive bird-dispersed species such as *B. thunbergii*.

INTRODUCTION

In the last five years, at least four studies have investigated the spatial distribution of the invasive shrub *Berberis thunbergii* in the E. S. George Reserve. Each study used different methods to answer the same fundamental question: how is *B. thunbergii* arranging spatially and why? The George Reserve is 525 hectares and cannot be exhaustively sampled. Any investigation with finite resources must reconcile resolution with area of data collected. Each of the four studies settled on a different assay strategy and scale. This study summarizes past methods and presents new data on the finest-resolution analysis of *B. thunbergii* spatial distribution thus far.

*B. thunbergii* spatial studies at the E. S. Reserve have followed two geometric patterns: lines and squares. Both methods bring their strengths and weaknesses. Line-transects give fast and accurate percent cover data (Floyd and Anderson 1987), but are known to intersect more frequently with larger individuals or clumps than smaller ones, and therefore over-represent the presence of larger clumps (Drummer 1987). In 2005, a line-transect was used to estimate *B. thunbergii* patch size and percent cover in Big Woods (Vandermeer unpublished). This bias may explain the significant difference in average clump diameter from Vandermeer (unpublished) (1.88m) to data the clump diameter presented in this study (1.47m).

A square, quadrat, belt-transect, or other area-based method of data collection also can also provide percent cover, but is sensitive to over-estimation (Floyd and Anderson 1987). Furthermore, as the perimeter-to-area ratio increases for the area sampled, bigger clumps again become more likely to be recorded than smaller ones. In 2009, Grimm *et al.* visually estimated percent cover of *B. thunbergii* in 3m circles around hundreds of adult trees, sampling within a wide belt transect. This method may have also been biased. It has been shown that *B. thunbergii* may form more diffuse stands in areas with high canopy cover, such as beneath dense Witch Hazel groves (Guittar *et al.* 2010), or in forests that have not been logged in recent decades.
(DeGasperis and Motzkin 2007). Density measurements of *B. thunbergii* directly beneath tree trunks may have resulted in artificially low estimates. If adult trees were particularly close, measurements potentially led to double-counts.

This paper is the third study on *B. thunbergii* spatial distribution in the E. S. George Reserve in 2010 that has used modified area-based survey methods (Cable *et al.* 2010, Guittar *et al.* 2010). Cable *et al.* (2010) estimated 16 hectares of *B. thunbergii* cover at a 50m x 50m resolution. Guittar *et al.* (2010) used a similar method with 10x10m quadrats over 2 ha.

The results from previous studies (Grimm *et al.* 2009, Guittar *et al.* 2010) have found Japanese Barberry spatial distribution to follow a power-law distribution at larger scales (16 ha, 2 ha, 1.8 ha). This finding is significant because power-law distributions have been considered in many studies as a potential signature of self-organization (e.g. Zeide 1987). In plants, a power-law pattern in clump size vs. frequency might imply density dependence. Grimm *et al.* (2009) and Guittar *et al.* (2010) used percent cover in 2500m$^2$ and 100m$^2$ quadrats as a proxy for clump size, respectively. In this study we investigate clump size directly to see if the spatial organization and invasion patterns of Japanese Barberry are consistent at an even higher resolution (1m$^2$ over 0.5 ha). We corroborate previous findings that clump size vs. frequency roughly follows a power-law distribution, further strengthening the hypotheses that barberry exhibits a scale-free distribution. However, we acknowledge the fact that phenomena other than self-organization can lead to an apparent power-law distribution. Thus, we also propose a simplistic theoretical model for exponential colonization and growth of an invasive bird-dispersed species such as *B. thunbergii*. Although we do not have sequential data to fit to the model, future studies may find it useful.

**METHODS**

I collected data in a 50x100 m plot in hectare “A” of the E. S George Big Woods forest plot (Figure 1). To provide reference, the study area was divided into 10m x 10m plots using transect tape and flags. I estimated the location of each barberry clump to the nearest 1x1m coordinate, and measured the major and minor radii with a meter stick. Barberry clumps overlapping the transect tape were noted and counted only once. All individuals were recorded regardless of size and radii were rounded to the nearest 0.25m.

Figure 1: The Big Woods plot of the E. S. George Reserve. The black square represents the surveyed area, from (0,100) to (50,200)
RESULTS

Spatial data

The 50x100m transect harbored 397 discrete *Berberis thunbergii* clumps (mean = 2.61 m²; median = 0.79m²) (Figure 2A). Total clump area for each 10x10m square was summed and graphically summarized in Figure 1B, and compared to visual estimate data from Guittar et al. in Figure 2C (Field Ecology 2010). To compare consistency between measuring and estimation methods, measured and summed data was ranked and plotted against estimated data (Fig 3). Consistent with past studies, size-frequency distribution roughly follows a power law (R² = 0.74)(Figure 6).

Figure 2: A (left) depicts 397 barberry clumps of sizes varying from 0.2m² to 50.3m². Clumps are arranged on a 1x1m matrix. B (center) is data from A summarized into 10x10m quadrats. C is data reported from Guittar et al. (2010) from the same area but recorded using visual estimation methods at a 10x10m resolution.
Theoretical Model of Exponential Growth

The theoretical model was calculated based on the following assumptions: after each time unit $t$, each circular clump radius $R$ grows by $a$ and reproduces successfully once. That is, $N(t) = 2^t$. Radial clump growth can represent clonal expansion and/or germination of a new individual.
immediately beneath its parent. Since the system is meant to represent a new but isolated invasion, the system is assumed to be closed and does not experience immigration, emigration, or a death rate. To simulate seed dispersion by birds, progeny are placed randomly in the plot. Random coordinates were calculated using a random number generator (http://www.random.org/) and plotted in Excel. Theoretical area can be calculated by the following equation:

Equation 1:

\[ A(t) = \pi \sum_{k=1}^{t-1} (t - k)(R + (k - 1)a)^2 \]

Forcing \( A(N) \) to fit to our data (\( A = 1040 \text{m}^2 \) and \( N = 397 \)) we can solve for \( a = 0.75 \text{m} \). Since \( A \) and \( N \) both grow exponentially they have a linear relationship (Figure 1B).

Figure 4: Area of total \( B. \) thunbergii as individuals (N) increase as predicted by equation 1. Blue diamonds are discrete theoretical points calculated at \( t=1\text{-}9 \). The red square represents the only measured data point at \( N=397 \), \( A=1040\text{m}^2 \).

Figure 5: A spatial depiction of one random invasion sequence as predicted by equation 1. From left to right: A (\( t = 2; N = 4 \)), B (\( t = 3; N = 8 \)), C (\( t = 7; N = 128 \)), D (\( t = 9; N = 512 \)). Points are randomly plotted and represent \( B. \) thunbergii clumps.
DISCUSSION

Spatial Data

Our results indicate that this study’s higher resolution method to measuring *Berberis thunbergii* spatial distribution is generally consistent with previous methods. When spatial data was summarized into 10x10m quadrats, *B. thunbergii* density correlated with estimated percent cover (Fig 2). Moreover, when percent cover data from both methods are compared data display variance but nearly identical trend lines (Fig 3; trend lines not shown). Consequently, we believe that data from the two methods is comparable at the 10x10m resolution.
Theoretical Model

The model reported in this study is overly simplistic and was used only to investigate the nature of an invasion that is growing exponentially in number and area covered. However, a surprising similarity between the predicted and measured size-frequency histograms (Figure 6) may suggest that the model is potentially relevant. That is, the number of smaller clumps is less than predicted (or conversely, the number of large clumps is more than predicted) by a strictly power-law relationship in both measured and predicted data. There are many possible alternative explanations for why this pattern might arise in empirical data. One is that small clumps mesh together to create indiscrete large clumps, simultaneously reducing the small clump number and raising the large clump number. Another mechanism could be stronger selection against small clumps. A final explanation could be experimental error when recording small plants. Nonetheless, the similar trend between the measured and predicted size-frequency distributions is intriguing and deserves further attention. Additional models have been explored in Hastings et al. (2005) and provide avenues of future theoretical research on the modeling the spread of invasives.

WORKS CITED


Vandermeer, J.Unpublished data.

HERBACEOUS UNDERSTORY DIVERSITY AS A FUNCTION OF UPPERCANOPY COVER AND AS A DETERMINANT OF TREE COMPETITIVE COMMUNITY DYNAMICS

SAHAR HAGHIGHAT

ABSTRACT

Tree competitive dynamics might be influenced by other trees as early as their emergence. Many times the understory flora and the effects that it might have upon the community dynamics of the uppcanopy are vastly overlooked. In this study the ground flora surrounding two tree species: red maple (Acer rubrum) and witch hazel (Hamamelis virginiana) were sampled; given their differences in height and tree crowns. Sixteen quadrats were surveyed to determine the herbaceous biodiversity established on the forest ground, under either species, while measuring crown cover through densitometer readings. It was assumed that tree species with lower canopies suppress light more than taller species and therefore it was hypothesized that trees of lower stature such as witch hazel would be stronger and would be a more effective light competitor than a taller species such as red maple. Shannon-Weiner indices and densitometer readings yielded no significant differences underneath either tree species.

INTRODUCTION

The understory herb layer is usually overlooked when studying forest dynamics. Many investigators discard the fact that the plants composing the understory and the microhabitat conditions upon which they are exposed to, might determine the species richness and composition in the future. The understory canopy only takes up about one percent of the biomass found in temperate forests (Gilliam 2007). Yet, despite the deceivingly low mass, the diversity characterizing it makes up roughly ninety percent of all forest plant species. The plants of this level are also significant contributors to the overall production of the forest- responsible for almost twenty percent of litter (Gilliam 2007, Muller 2003, Newfeld and Young 2003).

Taking into account all of these factors, it is not surprising to observe that there is a definite linkage between both upper and lower forest canopies and that this linkage is noteworthy in order to better understand trends in tree spatial competitive dynamics (Gillman and Roberts 2002). The composition of the herb species that are present in the lower canopy is affected by many different factors. One of which is the amount and type of light that reaches the herb layer. This is mostly determined by the tree species present in the upper canopy and the characteristics that define the tree crowns of these species present. Tree crowns and their structural shape determine how much light is blocked out and the different light wavelengths that are able to infiltrate to the forest floor.

With this in mind, it is assumed that shade tolerance will play a great role in the ability of various herbs and seedlings to emerge. Herbaceous flora, dependent upon its ability to persist given low light levels underneath dense tree crowns; will perhaps initially serve as tougher competitors towards seedlings that might tend sprout later on once these herbs have become established (Gilliam 2007). Light levels under varying tree crowns will create different microhabitats that will serve as appropriate fostering grounds for different species depending on
their ability to tolerate shade from upperstory trees as well as the herbs in their immediate surroundings that also serve as influential competitors.

In this study the ground flora surrounding two tree species were sampled: red maple (*Acer rubrum*) and witch hazel (*Hamamelis virginiana*). Tree species were selected given their differences in height and tree crown characteristics. It is assumed that tree species with lower canopies suppress light more than taller species and therefore it is hypothesized that trees of lower stature such as witch hazel will be a stronger and more effective light competitor than a taller species such as red maple. Maintaining these caveats in mind it thought that this might lead to a decrease in the biomass and diversity of the understory flora found underneath individuals of either species.

**METHODS AND MATERIALS**

*Sampling Methodology*

The sampling took place at the Big Woods plot of the ES George Reserve, on October 16, 2010. Sixteen quadrats were sampled based on their proximity, within a five meter radius, to either witch hazel or red maple. The exact location of the quadrat sampled was randomized. Each quadrat was measured off to 1 X 1 m$^2$. At each quadrat all herbaceous species were identified to species name - if possible, contrary they were assigned a morpho-species name and keyed out at a later time. The number of individuals or percent coverage was approximated for each species. Percent coverage was used to count individuals such as sedges or climbers, which were difficult to either determine individual plants from one another or count each individual separately - due to high volume. At every quadrat, densitometer readings were taken at the center of each quadrat. Percentage values were accounted for as individuals when calculating diversity indices.

*Statistical Analysis*

All statistical analysis were conducted on Excel and Estimate S. Importance values for each species were calculated using relative frequency and density. Shannnon-Weiner indices were calculated to estimate biodiversity under either witch hazel or red maple.

**RESULTS**

<table>
<thead>
<tr>
<th>spp name</th>
<th>Importance value</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Geum</em></td>
<td>0.146</td>
</tr>
<tr>
<td><em>Galium aparine</em></td>
<td>0.114</td>
</tr>
<tr>
<td><em>sedge</em></td>
<td>0.107</td>
</tr>
<tr>
<td><em>Berberis thumberghii</em></td>
<td>0.101</td>
</tr>
<tr>
<td><em>Geranium</em></td>
<td>0.091</td>
</tr>
<tr>
<td><em>Prunus serotina</em></td>
<td>0.079</td>
</tr>
<tr>
<td><em>Hamamelis virginiana</em></td>
<td>0.067</td>
</tr>
<tr>
<td><em>Aesclepius quadrifolia</em></td>
<td>0.057</td>
</tr>
<tr>
<td><em>Prunus serotina</em></td>
<td>0.057</td>
</tr>
<tr>
<td><em>Toxicodendron radicans</em></td>
<td>0.045</td>
</tr>
<tr>
<td><em>A. petiolata</em></td>
<td>0.034</td>
</tr>
<tr>
<td><em>R. americanum</em></td>
<td>0.034</td>
</tr>
<tr>
<td><em>Fraxinus</em></td>
<td>0.034</td>
</tr>
<tr>
<td><em>Viburnum lentago</em></td>
<td>0.034</td>
</tr>
</tbody>
</table>
Table 2: Importance values for all species underneath witch hazel

<table>
<thead>
<tr>
<th>spp name</th>
<th>Importance value</th>
</tr>
</thead>
<tbody>
<tr>
<td>R. allegheniensis</td>
<td>0.175</td>
</tr>
<tr>
<td>Hamamelis virginiana</td>
<td>0.123</td>
</tr>
<tr>
<td>sedge</td>
<td>0.116</td>
</tr>
<tr>
<td>Allaria petiolata</td>
<td>0.110</td>
</tr>
<tr>
<td>Geum</td>
<td>0.096</td>
</tr>
<tr>
<td>Barberis thumberghii</td>
<td>0.092</td>
</tr>
<tr>
<td>Prunus serotina</td>
<td>0.061</td>
</tr>
<tr>
<td>Veronica serpyllifolia</td>
<td>0.057</td>
</tr>
<tr>
<td>herb</td>
<td>0.039</td>
</tr>
<tr>
<td>Viburnum lentago</td>
<td>0.039</td>
</tr>
<tr>
<td>Galium aparine</td>
<td>0.031</td>
</tr>
<tr>
<td>Acer rubrum</td>
<td>0.031</td>
</tr>
<tr>
<td>Prunus serotina</td>
<td>0.031</td>
</tr>
</tbody>
</table>

Figure 1: Species accumulation curve for red maple
Table 1 & 2 show the relative similarity in biodiversity of herbaceous species underneath either tree species. Figures 1 & 2 are species accumulation curves that show the comparative similarity in species richness under both tree species. The curves indicate that both microhabitats underneath either tree species canopies possess a similar variety and abundance in species compositions. This was further confirmed with Shannon-Weiner indices, non-significant, indicating 2.437 for red maple and 2.225 for witch hazel. Both tree species seemed to harbor similar species. Both tree species shared seven species in common with red maple having a total of fourteen species compared to a similar thirteen species under witch hazel trees. Densiometer indicate 52% under red maple and 50% under witch hazel. There was no significant difference amongst the canopy cover for either tree species.

DISCUSSION

Biodiversity is dependent upon many different factors that are very difficult to disassociate from one another. The microhabitat that seedlings or herbs are initially exposed to when they first emerge is critical to their survival. In previous studies it was determined that there is a link between the lower and upper levels of temperate forests. The lower level, herbaceous canopy, is completely dependent upon the habitat conditions that overarching trees impose upon them. Light levels vary according to the coverage that different trees shade upon the forest floor and therefore upon other forest flora. Characteristics defining the shape, density, and overall coverage of tree species contribute to their immediate forest floor microhabitats by varying the amount of light reaching the ground as well as the type of light, amongst other factors not investigated in this survey.

In this study it was found that there was no significant difference in either the canopy cover or the biodiversity of species underneath the canopies of red maple or witch hazel. It must
be noted, however, that the sampling for this study was conducted well into autumn, past the first frost of the year. All the trees sampled had lost the vast majority of their leaves and much of the understory flora had been affected greatly by the frost. Given the constraint, much of the data collected by the densitometer was insufficient to make an accurate approximation of the possible light levels that might infiltrate to the ground when either tree species is in full bloom.

For future samplings or in continuation with a similar investigation, it is greatly advised that precaution be taken to conduct the survey earlier in the season or at least before it begins to frost. Also, different tree species should be investigated, especially those with greatly varying heights and crown shape - in order to see more marked differences amongst the effects of the species investigated.

ACKNOWLEDGEMENTS

Thanks to Dave Allen and John Vandermeer for their patience, advice, and humor.

LITERATURE CITED

Gilliam FS. 2007. The ecological significance of the herbaceous layer in temperate forest ecosystems. BioScience 57:10 845-858


DO POLLINATORS PREFER LATE-BLOOMERS? LATE BLOOMING FLOWERS
OF THE E.S. GEORGE RESERVE AND THEIR ABILITIES TO ATTRACT
POLLINATORS

ABIGAIL HYDUKE

ABSTRACT

Late-blooming or fall-blooming is a strategy that can provide plants with
fewer competitors for attention by pollinators. If it is the only time of year
that a plant flowers, it is a gamble that pollinators will also be actively
foraging late in the season. Some species may increase their visibility and
attractiveness to pollinators by large floral displays. However, once a
pollinator is initially drawn in, it may frequent nearby less flamboyant
blooms which we assume to possess equal rewards.
I predicted that species with milder displays close to species with prolific
flowering would receive benefits from their proximity with fewer initial
investments.
My results imply that species with larger floral displays receive the most
attention by pollinators, but because there was only one species blooming
abundantly during the study this may not account for additional pollinator
preferences.

INTRODUCTION

Pollinating insects are attracted to flowers by a variety of mechanisms including
color, size, shape, odor, nectar-guides and even simulated pheromones (Muller 1883,
Anders Nilsson 1992). The initial draw may provide them with one of many rewards or
as in the case of some orchids simply be a deceptive device to obtain pollinator services
relationships may be very specific, in that only one pollinator will lead to viable
offspring, or they can be more general, such as flowers that are receptive to a variety of
pollinators (Fenster 2004).

Flowering patterns can be described as a matter of timing, duration and frequency.
In an effort to limit competition for pollinator attention and seed dispersal, plants may
prefer to extend their flowering times or even bloom in the late summer or fall (Bawa
1983). Mass blooming is another strategy used by some plants to entice pollinators with
large floral displays (de Jong 1992). However, if a plant invests all of its floral screen
time for a late viewing, it is gambling that pollinators will still be foraging at the same
time.

Invasive species are largely successful because of their adaptability. Many
important invasive plant species in North America flower prolifically (Lythrum salicaria,
Centaurea maculosa, Cystis scoparius) and are receptive to a variety of pollinators
(Brown 2002).
While the pollinator may initially be drawn in by species which invest energy into mass-flowerings, others able to co-occur will receive residual benefits as pollination is facilitated towards a display that would otherwise have been overlooked (Nielson, 2008).

I observed pollination events as they occurred on flowers in an attempt to determine what qualities of a late-bloomer were most successful at attracting pollinators. I predicted that species with large floral displays would initially draw in pollinators, but that once captive, the pollinator would also forage on species with fewer blooms.

METHODS

I selected 10 1x1m² sites within the Oldfield area of The E.S. George Reserve in Livingston County, Michigan. I selected the sites according to their composition of blooming species.

Starting at 11am on Sunday, October 17, 2010 I spent 20 minutes regarding the flowering plants within each site and recording pollination events. Once a pollinator landed on a clump I did not record visits to flowers of the same species within that clump unless it clearly left and returned in which case I could not distinguish it from a pioneer pollinator. If pollinators went to multiple species of blooms within the sites, I recorded the shift as a separate pollination event. I also estimated height and percent coverage for blooming plant species within the designated sites.

RESULTS

Table 1. Late-blooming species observed in sites.

<table>
<thead>
<tr>
<th>Code</th>
<th>Scientific Name</th>
<th>Common names</th>
</tr>
</thead>
<tbody>
<tr>
<td>CEMA*</td>
<td>Centaurea maculosa</td>
<td>spotted knapweed</td>
</tr>
<tr>
<td>SYPI</td>
<td>Symphyotrichum pilosum</td>
<td>hairy white oldfield aster</td>
</tr>
<tr>
<td>SYOO</td>
<td>Symphyotrichum oolentangiense</td>
<td>sky-blue aster</td>
</tr>
<tr>
<td>SOAL</td>
<td>Solidago altissima</td>
<td>late goldenrod</td>
</tr>
<tr>
<td>SORI</td>
<td>Solidago rigida</td>
<td>stiff goldenrod</td>
</tr>
<tr>
<td>DACA*</td>
<td>Daucus carota</td>
<td>Queen Anne’s lace</td>
</tr>
</tbody>
</table>

*non-native and potentially invasive species
Figure 1. Sites described by their species and their percent coverage

*Symphyotrichum pilosum* had the highest percent coverage of any species by site. There were no other late-blooming species available with comparable displays. I made an effort to select sites with no *S. pilosum* for a comparison.

Figure 2. Number of visits by plot per species.

Figure 3 shows that pollinator preference was focused on *Symphyotrichum pilosum*. Pollinators were only observed on *S. pilosum, Centaurea maculosa* and *Daucus carota* and ignored several sites with other species cover.
There is no relationship between the frequency of pollination events and height. There was a positive correlation between species percent coverage and visits by pollinators. Of the total pollination events that occurred, 58 happened on the abundant species of *Symphyotrichum pilosum*, three occurred on *Centuarea maculosa* and one event was observed on *Daucus carota*. 

*Figure 3. Pollinator frequency and height of flowering species*

*Figure 4. Visits by pollinators according to percent coverage.*
I divided pollinators into 4 different categories of bumblebee, fly, honeybee and ant. Bumblebees were the most abundant pollinators, followed by honeybees and flies (which consisted of at least 4 types of flies) and lastly a single ant foraging.

**DISCUSSION**

My results indicate that pollinators were most attracted to species with the largest floral display. In this case *Symphyotrichum pilosum* was the only species observed within the old-field to possess a cover greater than 5%. While I selected sites according to composition, the pollinators did not and foraged almost exclusively on *S. pilosum* when it
was present. Of the four types of pollinators observed (Figure 5), other than the singular ant incidence, the preference was dominated by *S. pilosum*.

Figure 4 denotes a highly positive correlation between cover and pollination event. But without another species to compare this across, I cannot confidently say it is not another characteristic of *S. pilosum* which not only attracts pollinators but continues maintain their attention once they enter the flowering clumps. Pollinators may also have been responding to the white-color of the ray-flowers or the prospect of rewards available in this species over its late-blooming cohorts.

The two invasive species in this study (*Centaurea maculosa* and *Daucus carota*) did not receive much attention. At their current flowering status they cannot be held culpable for distracting pollinators from providing services to native species. Yet, if both species exhibit large floral displays at a different time of year, which is likely given the observed abundance of seed-heads around the remaining flowers, they might possess a competitive edge. I presume that the majority of flowering had already occurred in these species and a few remaining flowers were an additional investment, but not integral to the plants survival.

The foraging insects preference for the most abundant bloom may also be related to its short term memory and inability to recognize other species with equal rewards. Flower constancy is beneficial to conspecifics, but less so for insects which are bypassing alternatives. The tendency of pollinators to focus on the most abundant species may be related to the formation of a search image with limited short-term memory capacity. (Chittka et al. 1999).

As the only species blooming abundantly at the time of this study, the energy and gamble invested in a mass-blooming event appear profitable to *S. pilosum*. Investigating fertilization rates and seed viability are additional measures that would contribute towards quantifying the true value of blooming patterns.

ACKNOWLEDGEMENTS

Dave Allen, Ivette Perfecto, John Vandemeer and Field Ecology 2010

REFERENCES


AN INVESTIGATION OF AUTUMN MAPLE LEAF COLOR IN E.S. GEORGE RESERVE
MINGQI LIU

ABSTRACT

Autumn leaf color is determined by the distribution of three leaf color pigments. Anthocyanin pigment which is synthesized before leaf senescence has significant functional roles as photoprotection or anti-herbivore. In this study, I examined the relationship of leaf color ratios of three maple species with sunlight exposure – which is an important indicator of whether a leaf needs photoprotection. I found strong correlation of leaf color ratio with canopy cover. North-faced branches have more green leaves than others. And with different species, silver maple tends to have higher red leaf color ratio than the other two. Seedlings, which have higher canopy cover had much higher green leaf ratio than adult trees. A single leaf from a tree cannot predict the ratio of the whole tree.

INTRODUCTION

The deciduous forests in North American are most renowned for their brilliant and diverse foliar colors when fall came (Schaberg et al., 2008). The mosaic of leaf colors during this time was determined by variation in three tree plant pigments: chlorophylls – the green color, carotenoid – the yellow color and anthocyanin-the red color. Chlorophylls and carotenoid are synthesized during the growing season for photosynthesis (Taiz and Zeiger, 2002). But synthesizing anthocyanin is more energy consuming and it is of the synthesized shortly before leaf shedding (Matile, 2000; Hoch et al., 2001; Lee, 2002). But why trees would spend energy on leaves that are nearing senescence?

Traditionally, it is believed that anthocyanins are just a by-product of leaf senescence (Archetti, 2000; Matile, 2000). Recently, however, evolutionally value of autumn colors were discovered and more than ten hypothesis were put forward to explain that (Archetti, 2009). Among them, two are the major theories: 1) the photoprotection theory –anthocyanins act as either red sunscreen to protect plants against photoinhibition or act as anti-oxidant to protect plants against photooxidation. Anthocynin can soak up radiant energy at wavelengths poorly absorbed at other accessory pigments, such as in the green waveband (Lee and Gould, 2002) and absorb extra free radicals that can destroy the biological machinery around them which are potentially damage the leaves (Lee and Gould, 2002); 2) the anti-herbivory theory, like a coevolutionary hypothesis which concern aphid-tree relationships (Wilkinson et al., 2002; Schaefer and Rolshausen, 2006, 2007). A third theory is the combination of the former two.

Many factors can influence red color expression in tree leaves. Previously researches have provided evidence supporting their potential value in facilitating protracted nutrient recovery during leaf senescence, and leaves with higher N concentration would turn later than leaves (Field et al., 2001; Hoch et al., 2001;
Schaberg et al., 2008). Other factors, such as cool temperature, would also influence anthocyanin pigment synthesizing.

Among them, two important factors cannot be ignored: one is exposure to sunlight and the other is freezing. The intensity and length of sunlight exposure would be very important to determine anthocyanin synthesize according to photoprotection theory. Freezing would cause leaves to drop earlier before they have a chance to develop fall coloration (Aker et al., 2009).

Maple leaves are renowned for their various coloration in autumn (Schaberg et al., 2008). There are three maple species, sugar maple (A. *Saccharum*), red maple (A. *rubrum*) and silver maple (A. *saccharium*) that grow in different habitat types in E. S. George Reserve. In this study, I am going to examine the relationship of maple leaf coloration with sunlight exposure. My hypotheses are 1) the more open site should have higher of red leaf ratio; 2) the percentage of red should show positive correlation with canopy cover; 3) the green leaves should be remain in north-faced branches; 4) the trees with more than 50% leaves remaining on the branches should have more green and yellow leaves and 5) seedlings have more green leaves and yellow leaves than adult trees.

**METHOD**

*Study site*

I have three study sites, which are located in the E.S. George Reserve, Ann Arbor, Michigan (approximately 42°28′N, 84°00′E). Site1 was located in West Wood, dominated by one very big *Acer Saccharum* (Sugar Maple) and several of its siblings. Site 2 was located in Big Wood, where the forest canopy is largely dominated by *Carya* (Hickories), *Quercus* (Oaks), and to a growing extent *Acer rubrum* (Red Maple) and *Prunus serotina* (Black Cherry). Site3 was located near crane pond, along the road that separated the two ponds.
Sampling

I randomly selected the maple trees in all the sites. For Site 1 and Site 3, I measured the DBH of each tree and for Site 2, I wrote down the number of the tag and looked for the DBH from the Big Woods database (Allen, unpublished data). I also estimated the height of each tree. For the color of the leaves, I recorded the percentage of green (%), red(%), yellow(%) and brown(%) of entire tree. I also classified the canopy coverage of each tree into four categories: 0%-24.9%, 25%-49.9%, 50%-74.9% and 75%-100% canopy covered. I classified the trees as with more than 50% of their leaves fallen or less than 50% of their leaves fallen. For branches with green leaves, I faced the direction of the branch with most green leaves. I did the similar measurement for seedlings without recording their DBH. And I picked one representative leaf from each tree and photographed it in a white paper background using Cannon Digital camera.

Leaf color analysis

Images were imported into ImageJ and color ratios were calculated followed Zemenick et al., (2010).

Data analysis

All the data was put into Microsoft Excel 2007. The statistical tests and correlation were done in SPSS (version 13.0).
RESULTS

Sites and leaf colors

I found no significant difference between Site 1 and Site 2 or between Site 1 and Site 3 of either yellow, red and brown ratios (p>0.05). However, green ratio of Site 2 was lower than either Site 1 or Site 3 (p=0.017 and p=0.003, respectively; see Fig. 2).

Figure 2. Leaf color in different sites.

Species and leaf colors

There was no significance difference of yellow, red, green and brown ratios of different maple species (Fig.3). However, silver maple had more red leaf ratio and less yellow leaf ratio in comparison to the other two.

Figure 3. Leaf Colors in Different Maple Species.
Direction and green leaves

Of the nine trees that had obviously red, yellow and green part of the tree, only three of the total nine samples had most part of green leaves on south-faced branches (Fig. 4). The different ratios were just not let the points cover each other.

Figure 4. Green Leaves of Branches Faced Different Directions.

Canopy cover and leaf colors

For all the trees, there were significant differences between canopy cover for yellow (See Fig. 5). There was a trend that the ratio of green leaves increased with canopy cover, however, it is not statistically significant ($t=-2.58; p=0.058$ between 0-24 and 75-100).

Figure 5. Leaf Colors in Different Maple Species.
Leaf shedding and leaf colors

There were no significant differences for red, yellow and brown ratios between trees that had more than 50% of their leaves fallen and less than 50% of their leaves fallen (p = 0.400; p = 0.962 and p = 0.137, respectively). But there was actually significant difference of green leaf ratio between the two categories (p=0.031).

Adult trees and seedlings

I found significant differences of red, green and brown color ratios between adult trees and seedlings (p = 0.007; p = 0.006 and p = 0.005). But there was no difference of yellow color ratio (p = 0.542). That was in accordance with the more canopy cover of seedlings.
Tree height and leaf colors

For trees of different height, there was a significant negative correlation of yellow leaf ratio ($r = -0.516$, $p = 0.002$; Fig.7a) and a significant positive correlation of red leaf ratio ($r = 0.479$, $p = 0.005$; Fig.7b). However, no significances were found of either green leaf ratio or brown leaf ratio ($r = 0.159$, $p = 0.378$; Fig.7c; $r = -0.06$, $p = 0.741$; Fig.7d). For different $\text{D}^2\text{H}$, I found no statistically significance of all the four color ratios ($r = 0.213$, $p = -0.223$; $r = 0.110$, $p = 0.541$; $r = 0.146$, $p = 0.417$; $r=0.09$, $p = 0.619$).
Figure 7. Leaf Colors in Adult trees and Seedlings.

Yellow

\[ y = -1.4755x + 69.203 \]

\[ R = -0.516 \]

Red

\[ y = 1.3489x + 5.6852 \]

\[ R^2 = 0.229 \]
Can a single leaf predict the whole tree?

The correlation between canopy cover of a tree and its representative leaf was as follows: only red and brown leaf ratio has a significant positive correlation ($r = 0.009$, $p = 0.523$), whereas not significances for both yellow leaf ratio ($r = 0.071$ and $p = 0.741$) and green leaf ratio ($r = 0.019$ and $p = 0.931$).

DISCUSSION

For all the maple species, although there were no significant of leaf ratios, silver maple had higher red leaf ratio than the other two, which was perhaps because silver maple is shade-intolerant species, and need more sunlight to growth. So they tend to grow in more open area with higher sunlight exposure. Sugar maple, in contrast, is shade-intolerant species that tend to grow as sub-canopy tree in more closed stands.

As for one single maple tree, north-faced branches tend to have higher green
ratios, whereas south faced branches have higher red ones. This is perhaps because south-faced branches receive more sunlight than the north-faced branches, thus produced more anthocyanins.

As with different canopy cover, the less canopy cover, the more sunlight the tree would receive and has higher red color ratio. And with higher canopy cover, maple species tend to have higher green leaves and may fall later than that with less canopy cover.

While in previous study, Gan (2008) argues that bigger tree would have better adaption and thus produce more red leaves, my study is not support that. My findings, however, were that tall trees would have higher red leaf ratio, but not big trees with higher biomass.

These findings could support the hypothesis that higher sun exposure could induce higher anthocyanin production. Thus, this research is in support of the photoprotection theory.

As for the predicting ability of a single leaf for the whole tree, only brown and red color has such strong positive correlation. My results were consistently with Zemenick et al., (2010) that no single leaf has such prediction ability. That failure would perhaps due to the limiting of sampling size. But a more reasonable explanation would be that how much sunlight a leaf is exposed to is determined by its location on the tree and the relationship with other leaves on the same tree or the leaves on other trees that above it. So follow up study should sample more leaves on different part of a tree that can well represent all the leaves.

ACKNOWLEDGEMENTS

I would like to thank John Vandermeer and David Allen for their generally help in the field and David Allen, Ivette Perfecto and John Vandermeer for giving me invaluable guidance and suggestions for this project. I would also thank all the members in field ecology 2010 for your friendship.

LITERATURE CITED

Aker S., Bentz S., Clark E., Elias T. The Science of Color in Autumn Leaves. URL: http://www.usna.usda.gov/PhotoGallery/FallFoliage/ScienceFallColor.html.
Matile P. 2000. Biochemistry of Indian summer: physiology of autumnal leaf
A PREDICTION OF POTENTIAL DISPERSERS FOR ADHESIVE SEEDS
BINBIN LI

ABSTRACT
Although seeds that are dispersed by animals through epizoochory is largely ignored, it is the most efficient way for seeds to travel a long distance which is crucial for plant population and distribution. My study focused on investigating adhesive seeds diversity, height and adhesion ability for different animals. The first part of the study was in the field collecting seeds on pants according to different height categories. The second part was about attachment experiment on different specimen furs. The result showed that seeds outside structure and fur texture played an important role in the attachment process. Plants which have the same habitat preference with certain animals have higher adhesion ability for them. However, there seems no correlation between attachment success and animal height falls into the same height category as those plants.

INTRODUCTION
Seed dispersal plays a key role in determining the population dynamics and spatial distribution of plants (Nathan & Muller-Landau 2000). Seeds can be dispersed by animals via different ways. Recently most of the attention has focused on dispersal by frugivores, by ants and by scatter hoarding birds and mammals. The percentage of species containing adhesive seed which dispersed by means of animal fur and feathers is well known below 5% and dispersal of adhesive seeds has been largely neglected. However, adhesive fruits have the potential to travel farther than do seeds carried internally or dispersed by the wind (Harper 1977). This is because adhesive-dispersed seeds can be carried by the dispersal agent indefinitely as long as they do not detected and removed by the host animal. Dispersal for a long distance from parent plants is crucial for it provides plants with the opportunity to colonize new and distant habitats, and exchange genetic material between populations within metapopulations (Bohrer et al. 2005).

Unlike other animal-dispersed seeds, adhesive seeds do not provide dispersal agents with a nutritional or energetic "reward". They encounter with dispersal agents occur by chance (Sorensen, 1986). They usually do not display conspicuously outlook like fleshy fruits (Thompson and Willson, 1979). This results in the low attraction for animals to the parent plant. One of the explanations is that adhesive seeds usually are annoying for animals which would remove them as soon as they notice the adhesive seeds. Conspicuous color and odor will help animals to detect the presence of seeds quickly and avoiding visiting certain plants (Sorensen, 1986). Most of the adhesive species grow in the ground layer where possess more dispersal agents than the higher layer. One recent study on the Asteraceae family revealed that 87% of the adhesive Asteraceae species were low-growing, herbaceous plants (Venable and Levin, 1983).

There are two crucial steps in epizoochorous attachment and retention. The probability of seed attachment and retention is influenced by plant and seed traits such as outside structures, seed size, plant height (Couvreur et al. 2005a, Couvreur et al. 2008), the density and characteristics of the dispersal vector such as fur texture (Couvreur et al, 2005b; Mouissie et al. 2005).
My study investigated the biodiversity of adhesive seeds in E.S. George Reserve, seeds height and structures for different species. If plants height plays an important role in dispersal, could we predict potential disperser from their heights which are consistent with seeds? My hypothesis is that there will be higher attachment between seeds and animal furs that are in the same height category or prefer the same habitat.

**METHODS**

Five kinds of habitats in E.S. George Reserve such as forest, swamp, old field and transition area between forest and swamp, forest and old field were selected as my study areas (Fig.1). I conducted three 100-meter transects in each kind of habitat and collected all the seeds in my pants and regarded left and right legs as two repeats. Height was divided into seven category: 0-10cm, 10-20cm, 20-30cm, 30-40cm, 40-50cm, 50-60cm, >60cm. I marked the height category on towel and put it parallel to my pants when I sat down and collected seeds according to the division. Seeds were counted, sorted according to morphology and identified. Eight kinds of seeds were used for skin experiment to test their adherence ability. I selected five species, white-footed mouse (*Peromyscus leucopus*), chipmunk (*Tamias striatus*), skunk (*Mephitis mephitis*), raccoon (*Procyon lotor lotor*), white tail deer (*Odocoileus virginianus*) skin specimen from Natural History Research Museums which represent the most common species in different size category. I dropped seeds on the furs and then made it vertical to the table, waiting for 5 seconds. Percentage of seeds that still attached to the fur after 5 seconds was recorded. It was repeated three times for each specimen with each kind of seeds. ANOVA, Independent-samples T-test was used for data analysis.

![Fig. 1 Study area in E.S. George Reserve.](image-url)
RESULT

Diversity of Adhesive Seeds

13 species were found and except *Galium sp2* which was bigger than *Galium sp2* others were taken photos as follow (Fig.2). Except one species, the others were identified at least to genus. *Geum cf. virginianum* and *Hackellia sp1* were found in all of the habitats. *Asteraceae sp2, Galium sp2, Hackellia sp2,* and *Phryma leptostachya* only appeared in swamp area or in the swamp-forest transition area. *Geum cf. canadense* was mainly appeared in forest. *Galium sp1* was only found in old field. Moreover, *Desmodium cf. glutinosum* was found only in the edge of forest (Fig. 3).

![Fig.2 Photos for collected seeds](image)

Old field had the highest abundance of seeds while the richness was low. Swamp which had higher biodiversity according to calculated Shannon-Weiner Index had more species than forest and old field. Transition area tended to have more species found near the edge than in the center of forest (Fig.4).

![Fig.3 Comparison of richness and abundance in five habitats.](image)
Fig. 4 Species accumulation curves

Table 1. Shannon-Weiner Index

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Old field</th>
<th>Old field/forest</th>
<th>Forest</th>
<th>Forest/Swamp</th>
<th>Swamp</th>
</tr>
</thead>
<tbody>
<tr>
<td>Index</td>
<td>1.07</td>
<td>1.32</td>
<td>1.03</td>
<td>2.04</td>
<td>1.85</td>
</tr>
</tbody>
</table>

**Height of Seeds**

Different species assembled in different height (Fig. 5). *Asteraceae Bidens*, *Galium virginianum*, *Desmodium canadense* and *Geum canadense* distributed most in 0-10cm. *Galium sp2* and *Hackellia sp1* distributed most in 10-20 cm and *Geum virginianum* and *Hackellia sp2* were found mostly in 20-30cm. Less than 10 seeds of *Asteraceae sp2*, *Phryma leptostachya* were collected which distributed among 0-30cm. Rosaceae sp1 distributed among 20-40cm, while *Desmodium canadense* had two distribution peaks which were in 0-10cm and 30-40cm. However, none of them showed significant difference in height assembly (p>0.05).

Fig. 5 Height distribution for different species.
Adhesion ability
I added up the proportion of hooked seeds for five specimens to compare adherence ability for different seeds (Fig.6). Unknown sp1, Galium virginianum, Hackellia sp1, Geum canadense, Hackellia sp2 which have hooks had higher proportion attached to skins than Desmodium Canadense, Desmondium glutinosum and Geum virginianum which do not have hooks (p<0.05). Hackellia sp1 had significant higher ability than Geum virginianum and Desmondium glutinosum and Galium sp1 showed significant difference with Geum virginianum as well (p<0.05).

Fig. 6 The result for adhesion ability experiment

Five skins showed significant difference in their ability to carry seeds (p<0.05) (Fig. 7). Coarse fur such as deer and raccoon skins had higher ability to attach seeds than smooth fur such as white-footed mouse, chipmunk and skunk (p<0.05). But seeds did not show significant higher adhesion to animals in corresponding height category (p>0.05).

Raccoon which prefers forest and swamp habitat (Kurta, 1952) showed higher but not significant retention for Unknown sp1, Hackellia sp2, Geum virginianum, Geum canadense which were found mainly in swamp or forest than Desmodium Canadense and Galium virginianum which found mainly in old field (F=0.303, p>0.05). However, white-footed mouse which avoid field area (Kurta, 1952) when they become adults show higher retention for the seeds like Desmodium Canadense and Galium virginianum than others (F=0.001, p>0.05). Chipmunk which avoid swamp area (Kurta, 1952) showed less retention for Unknown sp1, Hackellia sp2 which were only found near swamp area than the others (F=8.503, p<0.05).
Table 2. Proportion of attached seeds for different species

<table>
<thead>
<tr>
<th>Species</th>
<th>White-footed mouse</th>
<th>chipmunk</th>
<th>skunk</th>
<th>raccoon</th>
<th>deer</th>
<th>Height (cm)</th>
<th>Habitat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Desmodium Canadense</td>
<td>0.60±0.05</td>
<td>0.52±0.07</td>
<td>0.52±0.11</td>
<td>0.43±0.06</td>
<td>0.74±0.14</td>
<td>0-10</td>
<td>Old field / Forest</td>
</tr>
<tr>
<td>Galium virginianum</td>
<td>0.68±0.21</td>
<td>0.63±0.06</td>
<td>0.41±0.06</td>
<td>0.67±0.10</td>
<td>0.68±0.10</td>
<td>0-10</td>
<td>Old field</td>
</tr>
<tr>
<td>Geum canadense</td>
<td>0.32±0.04</td>
<td>0.79±0.00</td>
<td>0.27±0.10</td>
<td>0.85±0.06</td>
<td>0.49±0.14</td>
<td>0-10</td>
<td>Forest</td>
</tr>
<tr>
<td>Hackellia sp1</td>
<td>0.66±0.16</td>
<td>0.64±0.02</td>
<td>0.66±0.02</td>
<td>0.6±0.05</td>
<td>0.60±0.07</td>
<td>10-20</td>
<td>All Swamp</td>
</tr>
<tr>
<td>Unknown sp1</td>
<td>0.54±0.09</td>
<td>0.58±0.08</td>
<td>0.56±0.11</td>
<td>0.58±0.18</td>
<td>0.83±0.13</td>
<td>20-30</td>
<td>Swamp / Forest</td>
</tr>
<tr>
<td>Hackellia sp2</td>
<td>0.28±0.00</td>
<td>0.15±0.00</td>
<td>0.33±0.25</td>
<td>0.85±0.00</td>
<td>0.78±0.08</td>
<td>20-30</td>
<td>Swamp / Forest</td>
</tr>
<tr>
<td>Geum virginianum</td>
<td>0.33±0.13</td>
<td>0.45±0.02</td>
<td>0.48±0.14</td>
<td>0.73±0.09</td>
<td>0.47±0.13</td>
<td>20-30</td>
<td>Both transitions</td>
</tr>
<tr>
<td>Desmodium glutinosum</td>
<td>0.59±0.10</td>
<td>0.55±0.02</td>
<td>0.3±0.04</td>
<td>0.56±0.13</td>
<td>0.52±0.17</td>
<td>30-40</td>
<td>Both transitions</td>
</tr>
</tbody>
</table>

Fig. 7 Retention ability for different furs
DISCUSSION

Chipmunk and skunk followed the hypothesis that animals and plants in same habitat tend to have higher attachment proportion. White-footed mouse showed opposite trends to keep seeds in same habitat. It may be an adaption to its life history. Juveniles are dispersed into field and come back to forest when they become adults (Kurta, 1952). The high adhesion ability could help plants in old field disperse into forest area with migration of white-footed mouse. As plants in certain habitat encounters the animals prefer the same habitat most, higher adhesion to those animals help them to reinforce their population in their habitats. However, more researches should be done to further support this hypothesis.

*Hackellia sp1*, which was found a relative high abundance in all the habitats, has a high density of hooks. From personal experience, it is the most difficult species to remove no matter from pants or animal furs. This high adhesion ability may be one of the causes for its large distribution.

The study did not support the hypothesis that animals and seeds fall into the same height category will have higher attachment to each other. The texture of fur and whether seeds have hooks appendages seem to have much more crucial role in the proportion of attachment. It is consistent with the opinion Fischer *et al.* (1996) expressed that adhesive structures such as hooks, stiff hairs and awns, can increase seed attachment and retention in animal fur. The seeds without hooks such as *Desmodium canadense, Desmodium glutinosum*, and *Geum virginianum* tend to have flat surface with some short hairs which enlarge contact area with disperser.

Seeds in swamp and forest area are less abundant than in old field. It is because most of the adhesive plants are in ground layer which is highly restricted in forest and swamp for their higher canopy cover. But animals are more active and abundant in those areas. So the success of attachment relies on how many seeds could attach to an animal when it passes by. It could be deduced that large animals will have more seeds on its skin than small animals for it has larger area and stronger hit. Compared the size of animals, adhesive seeds would be much larger for a white-footed mouse than a raccoon, which in turn could be much easier for a mouse to detect its presence and remove it more quickly. So there may be a selective force favored the plants bearing their seeds higher in accordance to larger animals. More interesting raccoon and deer which are larger compared to the rest have higher proportion of attached seeds. Height, fur texture may reinforce each other’s effect during the dispersal process. However, reviewing my study, it showed that most of the seeds assembled in 0-20cm. I think it may results from the walking mechanism of humans. We lift our legs when we are walking, compared the size and relative stable movement of animals, this walking mechanism seems like a jumping fox dummy. So this survey method may influence the height estimation. To improve this experiment, researcher could push a animal dummy like the Hovstad *et al.* (2009) or just do vegetation survey to study how high seeds distribute for various species.

In conclusion, large animals with coarse texture of fur would have higher possibility to disperse adhesive seeds. Seeds with hooks could disperse a much longer way for they could have a high proportion attachment and hook tightly to the fur.
ACKNOLEDGEMENT
I appreciate the suggestion of John Vandermeer, the help of Philip Myers for skin experiment, and Robyn Burnham for identification of seeds. Also, I want to thank for David Allen for his support.

LITERATURE CITED


267
ABSTRACT

I studied the effect of temperature on competition and dominance in ant species. The focus was on the three most populous ant species in the field near the experimental ponds in the George Reserve: Solenopsis molesta, Monomorium minimum and the invasive Tetramorium caespitum. I tested the foraging and recruiting capabilities of the ants by baiting with tuna fish. The study was repeated on three different days with similar trends in affect of temperature on the ant species. The competition coefficients were calculated to determine the relationship of dominance between species over the food source.

INTRODUCTION

Many factors affect the dominance of a particular ant species. The factors include ability to find and secure food resources, fighting ability and aggressive behavior, and even pheromones used to pacify other species. How does temperature affect dominance between ant species? The physical environment affects foraging primarily through changes in humidity, soil moisture, daylight and temperature (Porter and Tschinkel 1987). The time of year a species forages depends upon the species. Prenolepis is known as a winter ant and has been reported to seen walking on the snow. Ants also have critical maximum and minimum temperature above or below which foraging does not occur (Bernstein 1979). For species that forage at the same time of year we want to know how does microclimate affect their foraging, recruiting and dominance for a food resource. Time of day foraging periods has been correlated with soil surface temperatures for many different species of ant (Bernstein 1979). This study examines the affect of temperature on dominance for Monomorium minimum, Solenopsis molesta and Tetramorium caespitum.

Past research determines that temperature does have an effect on foraging and recruitment for different ants species. Solenopsis molesta recruits at relatively high soil temperature, but forages day and night as long as temperatures are suitable (Vogt et al 2004). Monomorium minimum recruits at higher temperatures than sympatric ant species (Adams and Traniello 1981). Monomorium minimum does not forage at night (Claborn and Philips 1986). This study looks at the dominance of the three ant species in real time according to temperature.

Intransitive loops form between species when their competitive mechanisms keep each other in check. Specifically dealing with three different species, the dominance of one species over another, allows for a third which dominates one of the first two. Intransitive loops result in a constant rotation of species distributions, wherein each species is driving out an inferior competitor as it is being driven out by a superior competitor (Maloney et al 2010). This is comparable to the game of rock, scissors, paper. Each one beats one of the others and so you are left with a cyclical movement of dominance. Going back to ants, the species that form these loops keep each other in check in terms of their competitive strategies by harboring species that may not survive on their own ‘satellite species’ without the protection of ‘mosaic species’; yet these loops still tend to drive out other surrounding species that have to compete with these.
mosaic formations (Yitbarek & Vandermeer, unpublished 2010). The relationship between species can be examined in order to understand their interaction and dominance in a specific area. "Maloney et al. (2010) found an intransitive loop between these three species." I am investigating whether the intransitive loop is maintained at different temperatures. The competition coefficient is calculated for *S. molesta*, *M. minimum*, and *T. caespitum* in order to determine if an intransitive loop is present and to the degree in which temperature forms or breaks up the intransitive loop.

**METHOD AND MATERIALS**

**Study Site**

The study site was a fairly homogeneous 90x30 meters section of land located at near the experimental ponds in the E.S. George reserve. This relatively flat area was created by heavy machinery several years ago and it is characterized with a grassland cover and it has a rich insect community. Day 1 was conducted on October 9th, Day 2 was conducted on October 10th and Day 4 was conducted on October 12th.

**Data Collection**

This study focused on three ant species found in the uniform area: *Solenopsis molesta*, *Monomorium minimum* and *Tetramorium caespitum*. *S. molesta* and *M. minimum* are a native species to the United States while *T. caespitum* is an invasive species introduced from Europe. Five sites were chosen from a map showing location of dominant species. Two sites were chosen that are characterized as mixed dominance of all the species and then three sites characterized as dominant for each species. Ten tuna fish baits were placed 10 cm apart in a half circle for each site and the baits were marked with a flag. Air and ground temperature was taken at the start of each round. Each site was visited and the number of individual ants was recorded for each species at each of the baits. This was continued until the first bait was reached again whereas it was started over. This allowed about half and hour between rounds for each bait.

**RESULTS**

At the start of each round I recorded the number of individuals per species for all the baits. I used this data to calculate the competition coefficient. I recorded which species displaced another species during the round or the species with the highest number of individuals at the end of the round. I totaled this numbers for each of the species interaction. For example, I totaled how many times *Solenopsis molesta* dominated or displaced *Tetramorium caespitum*. I totaled these numbers for each bait on each site. Then I divided that number over the total number of baits *Solenopsis molesta* was present (Fig. 2). A higher competition coefficient indicates a stronger dominance species A has over species B. The competition coefficients are compared to see the direction of the dominance for species A against species B (fig. 2). The competition coefficients do not show an intransitive loop for day 1, 2 or 4. *Tetramorium caespitum* has a strong competition coefficient for both *Solenopsis molesta* and *Monomorium minimum* on each day. On the first day *M. minimum* dominates *S. molesta* but on day 2 and 4 the dominance is reversed.
Figure 2. This figure demonstrates the direction and strength of the competition coefficient.

Day 1
Average Temp
26 Celsius

Day 2
Average Temp
33 Celsius

Day 4
Average Temp
18.5 Celsius

For each day the total number of ants for each species was totaled at each temperature for each round (Fig. 3 & 4). Interestingly there is a close correlation between both days. The slope of the line was calculated for each species. On day 1 the slope for *T. caespitum* was 11.582, for *S. molesta* was 8.0113 and for *M. minimum* was 4.7581. On day 2 the slopes were higher but consistent with day 1. The slope for *T. caespitum* was 21.611, for *S. molesta* was 19.305 and for *M. minimum* was 7.9917. The slope represents approximately how many ants will increase with temperature for every 500 baits.

Figure 2 & 3. The two graphs show the total number of ants per temperature for all five baits both days.
This study does confirm that temperature has an affect on ant species, which is supported by similar research, however it does not show intransitive loops between the three species. The competition coefficients show that T. caespitum has a strong dominance over both S. molesta and M. minimum. This is due to perhaps the recruitment ability show in figure 3 & 4. T. caespitum has far greater numbers of ants present at each temperature than the other two species. Even if T. caespitum does not get to the food resource first it has the ability to exclude the other species by recruiting quickly and pushing them out. S. molesta and M. minimum also respond to the rise in temperature. T. caespitum is labeled as an invasive. And this study supports that assumption because a T. caespitum has a higher competition coefficient and more individuals at any given temperature. The graphs do not take into consideration the lapse in time with the rise in temperature. A trend is seen in the slopes in the graphs for day 1 and day 4, which supports the consistency of the data. The competition coefficients show a change of dominance from day 1 to day 2 for S. molesta and M. minimum. This could indicate coexistence between the two species. The reason for the change in direction could depend on different factors. The two species were observed occupying the same baits by foraging at different spaces.

It is difficult to calculate competition based on one factor. Temperature is not the only factor that plays a role in competition. It is hard to measure to what degree temperature has an affect without considering other factors. The species M. minimum releases a chemical interferences that delays invasion by competitors and allows more time for the colony to consume the food resource (Adams and Traniello 1981). The data did not show a high competition coefficient for M. minimum but instead this ability may allow the species to coexist with the other ant species. Another factor that could have influenced competition in ant species is where the baits were placed. Prior research mapped out the research area with dominance of the different species. Perhaps T. caespitum was present in one site over another because more T. caespitum nests were located closer to the baits. Because the same test sites were used for all
three days could have another affect on dominance. Although, the baits were picked up from day 1 to day 2 and then from day 2 to day 4, the ant species could have stayed close to the baits overnight and then to day 4.

Inefficiencies in collecting data could also cause the lack of intransitive loops. The degree of exclusion was not considered when calculating the competition coefficients. For example if *T. caespitum* was dominating the bait and if one individual of *S. molesta* was present it was counted as a takeover. Perhaps not as much weight should be given to this exclusion than if one or two individuals were able to push out a larger number of another species. Although, a large amount of data was collected only a small percentage of baits had a direct species interaction. Numerous baits were only occupied by one species and site 5 did not provide any data due to the lack of interspecies competition. More specific calculation of the competition coefficient could help to project a more accurate depiction of the competition between species. Further research ideas include observing the species interaction more closely on a limited number of baits spaced widely apart.

WORKS CITED


Claborn, D. and Philips, S Temporal Foraging Activities of Solenopsis invicta (Hymenoptera: Formicidae) and Other Predominant Ants of Central Texas. *The Southwest Naturalist*. 31:555


BERBERIS THUNBERGII AND ELAEAGNUS UMBELLATA IN THE FIELD AND FOREST

KASSANDRA C. SEMRAU

Abstract: The E.S. George Reserve provides a unique opportunity in which to study invasive species due to its historical use as an agricultural site and the infrequent disturbance it has experienced since it was acquired by the university. This study examines the abundances of two invasive species, Berberis thunbergii and Elaeagnus umbellata, along environmental gradients between forest and field habitats. Both species were expected to be more abundant along disturbed field edges but were actually more abundant within forest interiors.

INTRODUCTION

Factors affecting the spread of invasive species are not well understood. While attempts have been made to determine a set of characteristics that define what makes a nonnative species invasive, they have largely been unsuccessful because a variety of characteristics affect how well the species spreads and becomes established (DeGasperis & Motzkin, 2007). High sunlight availability, fertile soils and high disturbance are conditions that are typically thought to favor invasive species (Huenneke et al, 1990, Meekins & McCarthy, 2001). Recently other factors such as land-use history have gained support as major predictors of invasion (Mosher et al, 2009). Land-use changes create opportunities for invasion and establishment of nonnative species (Hobbs, 2000, With, 2002) though some species may be more affected than others (Mosher et al, 2009). Recently, Berberis thunbergii, Japanese barberry, and Elaeagnus umbellata, autumn-olive, have been the focus of studies on invasive species in the eastern United States (Mosher et al, 2009, DeGasperis & Motzkin, 2007, Ibáñez et al, 2009). Berberis thunbergii, Japanese barberry, is known to be both a post-agricultural and long-term abandonment specialist (Mosher et al, 2009) and is thought to be a successful invader due to its high shade tolerance (Silander & Klepeis, 1999) and low mortality following establishment (Ehrenfeld, 1999). Elaeagnus umbellata grows in dense thickets in disturbed habitats such as old fields, riparian zones, and agricultural grasslands (Baer et al, 2006).

The primary predictors of invasion include land-use changes and canopy closure, and secondarily the distance to a vegetation edge (Mosher et al, 2009). Intact forested areas were found to resist invasion by a suite of nonnative woody species, including Berberis thunbergii and Elaeagnus umbellata, compared to abandoned agricultural fields (Mosher et al, 2009). This study reported that high to moderate light levels and the absence of human intervention, characterizing vegetation edges and abandoned fields, created favorable habitats for woody invasive plants (Mosher et al, 2009). Another study found Berberis thunbergii more frequently and in greater abundance in abandoned agricultural fields than in continuously forested habitats (DeGasperis & Motzkin, 2007). In yet another study, smaller forested patches tended to be more easily invaded, implying the importance of the forest edge as a buffer zone in preventing the expansion of invasive species (Cadenasso & Pickett, 2001, Mosher et al, 2009).

Elaeagnus umbellata is present in both forest and field sites within the E.S. George Reserve, while Berberis thunbergii has been noted for its clonal expansion in the understory of the Big
It is not known how abundant *B. thunbergii* is within field sites at the E.S. George Reserve. This study will attempt to determine the abundances of two invasive species, *B. thunbergii* and *E. umbellata* along environmental gradients between two habitats, field and forest. Major questions addressed in this study include: Do *E. umbellata* and *B. thunbergii* competitively exclude one another? Do these species occur more frequently in abandoned agricultural fields than forested areas? Is abundance positively correlated with disturbance or sunlight availability? Does species abundance increase with proximity to vegetation edge (road)?

**MATERIALS AND METHODS**

Three sites were chosen in areas where the road separated open field from forest. These sites include one site at the Airfield grading into the forest surrounding Buck Hollow, and sites on the east and west end of Evans Old Field (Fig.1). Twelve 10 m x 10 m plots were chosen at each site, six in each habitat (field or forest), three of which were in close proximity to the road and three located further into the habitat and away from the road. Plots closer to the road were considered more disturbed than plots in the interior of each site. Within each plot an estimate of the percent light availability was recorded along with the number of *Elaeagnus umbellata* and *Berberis thunbergii* individuals. Only individuals more than 1 m away from conspecifics were counted, and there were no restrictions on the size of the individual. Other dominant species were noted for their presence at each site.

**RESULTS**

Paired t-tests showed that differences in abundance of *E. umbellata* between the field and forest was significant (p=0.00153) with an average of 0.39 individuals per field plot and 3.33 individuals per forest plot. The differences in abundance of *B. thunbergii* between the field and forest was also significant (p=0.0196) with an average of 4.11 individuals per field plot and 7.39 individuals per forest plot. Abundance of *E. umbellata* and *B. thunbergii* plotted against one another showed a slightly positive trend (Fig. 2). Abundance of *E. umbellata* and *B. thunbergii* plotted against percent light available led to a slightly negative correlation (Fig. 3). As distance to the road increased, the relative abundance of *E. umbellata* increased, but not as much as the relative abundance of *B. thunbergii* increased (Fig. 4).
Figure 2. Relative abundance of *B. thunbergii* and *E. umbellata* per plot showing positive association between the two species.

![Graph showing positive association between B. thunbergii and E. umbellata](image)

\[ y = 0.2085x + 0.6623 \]
\[ R^2 = 0.07873 \]

Figure 3. Relative abundance of *B. thunbergii* and *E. umbellata* against light availability, showing slightly negative correlation.

![Graph showing slightly negative correlation between B. thunbergii and E. umbellata](image)

\[ y = -0.0399x + 4.2449 \]
\[ R^2 = 0.19425 \]

\[ y = -0.0592x + 9.2865 \]
\[ R^2 = 0.23604 \]
DISCUSSION

Based on the results, *E. umbellata* and *B. thunbergii* are not acting according to the behavior exhibited in previous studies. Invasive species tend to occur in disturbed habitats, but in this study higher abundances of both species were found in plots near the interior of each site compared to those plots closer to the road. In one study, *B. thunbergii* was found to occur more frequently in undisturbed mesic sites than drier disturbed sites (Searcy et al, 2006), supporting the results from this study that the highest proportion of *B. thunbergii* occurred in plots in the interior of the forests. The lowest frequencies of both species were found in field plots and particularly those in close proximity to the road, suggesting that abandoned fields are not preferential habitats for these species, despite the characterization of *B. thunbergii* as a post-agricultural specialist. One possible reason for the low frequency of *E. umbellata* and *B. thunbergii* in the field is that nonnative species may be unable to establish due to the fact that physical space and competitive exclusion restrict the number of individuals able to persist (Levine, 2000, Davies et al, 2005). In the field, native grasses grow closely together and could exclude other species from establishing simply through competition for resources. Additionally, these two invasive species are primarily bird dispersed (Silander & Klepeis, 1999). In the interior of the field, there were few scattered junipers (*Juniperus sp.*) as the only site upon which a bird would be able to perch, with a slight increase with proximity to the road. It may be that there haven’t been enough opportunities for these invasive species to establish in the fields simply due to the infrequent dispersal events. In forested environments there are more opportunities to be dispersed by birds and less competition for space than in the field. Abundance of *B. thunbergii* in the forest increased rapidly with distance to the road and less light availability, hinting at its higher tolerance for shaded conditions than *E. umbellata*. Furthermore, once *B. thunbergii* has established, it is able to spread vegetatively, significantly increasing seed production without requiring a bird disperser to propagate. The ability of *B. thunbergii* to persist in highly shaded areas, to suffer low mortality, and to reproduce vegetatively are likely the factors responsible for
the increases in abundance and may be skewing the results of this study to favor forested habitats as optimal environments for these species. As these two species are both nitrogen-fixing shrubs, they share a niche and it could be postulated that they would competitively exclude one another. Interestingly, they were found to be positively correlated with one another. Bird dispersal may again be a driving factor in this correlation, though it could also be suggested that *E. umbellata* creates a semi-shaded habitat that *B. thunbergii* is able to thrive in. Future studies should focus on examining how closely in space these two species can coexist and to what degree bird-dispersal events play a role in their distribution.

Acknowledgements: Thanks to Robyn Burnham for aiding in the development of the field problem and its methodologies.

LITERATURE


LIANA RELAY ASCENSION AT THE E.S. GEORGE RESERVE

IMAN SYLVAIN

ABSTRACT
Lianas (woody vines) are important members of forest ecosystems primarily due to their negative interactions with trees. Lianas compete with trees for resources and use trees for structural support. Many vines are capable of co-inhabiting a single tree host. How or why the great abundance of lianas is supported by a single host is not well understood. I hypothesize resource partitioning through relay ascension is the key. Relay ascension is the process of one vine climbing an older vine on a host tree instead of the host directly. This may allow for smaller vines to ascend to the canopy tops where light-availability is optimal. The order in which different vines ascend trees may be determined by their climbing mechanisms. This study investigated the occurrence of relay ascension on the E.S. George Reserve. The width and climbing mechanisms of primary, secondary, and final vines on host trees were recorded. Whether young vines intersected or directly overlapped primary vines was noted to decipher whether relay ascension is formulaic or more prevalent for certain climbing mechanisms. Results of this study show that lianas with adventitious roots are first to colonize trees, and vines with adhesive tendrils arrive afterward. Lianas with adventitious roots are more often intersected and overlapped by other vines than lianas with adhesive tendrils. At the George Reserve relay ascension is commonly facilitated by lianas with adventitious roots, which are then climbed by lianas with adhesive tendrils. Relay ascension may provide a competitive edge for liana species which are unable to ascend large trees directly. By utilizing relay ascension lianas with adhesive tendrils can scale other vines in search of greater light availability or uncolonized canopy branches.

INTRODUCTION
The study of lianas (woody vines) is interesting for a variety of reasons. Approximately one-half of all vascular plant families contain climbing species. In fact, in some plant families, such as the Vitaceae and Hippocrataceae, nearly all species adopt climbing strategies (Putz 1984). Lianas are important components of both tropical and temperate forests and play significant roles in vegetation structure, productivity, and dynamics (Ichihashi et al. 2010). Lianas strongly compete with trees for above- and below-ground resources (Ladwig and Meiners 2010). In addition, lianas use trees for structural support as hosts. The relationship between lianas and trees can be viewed as parasitism (Stevens 1984), because lianas can cause great damage to the trees they inhabit. Lianas can cause trunk constriction and removal of bark, shoots, and buds on host trees (Ladwig and Meiners 2010). Lianas can also negatively affect the fecundity, survivorship, and growth of their host trees (Lutz 1943). In the Tropics, where lianas are so diverse and highly
abundant in forests, trees carrying lianas suffer higher mortality rates than trees without lianas and cause other trees to die upon falling (Putz 1984).

Lianas are well-known for their weedy growth habits (Ladwig and Meiners 2010). Their rapid growth may be facilitated by photo-response. Although most lianas are light-demanding and grow well in natural or man-made clearings (Putz 1984), some lianas are fairly shade-tolerant. Shade-tolerant lianas can wait for favorable light conditions, to which they respond with high growth rates (Ladwig and Meiners 2010). Areas of disturbance, particularly forest gaps and edges are colonized by lianas more quickly than trees, and generally have greater liana abundance. Previous studies on the E.S. George Reserve have shown that lianas may be gap pioneers and are most abundant within the first 5 meters of the road (Unruh 2009). Because lianas benefit from forest disturbance and fragmentation, their influence on gap-phase regeneration has negative implications for trees. Studies have shown that competition between lianas and shade-intolerant trees decreases diversity in forest gaps and stifles forest regeneration (Schnitzer and Carson 2010).

One possible reason why lianas are such strong competitors may be their ability to overwhelm trees. Variations in liana climbing mechanisms may play a role in the ability of liana species to overwhelm host trees (Remfert et al. 2009). A liana’s climbing is facilitated by the modifications of roots, stems, or leaves. Liana climbing mechanisms include adventitious roots, tendrils, touch-sensitive leaves, twining, and adhesive tendrils (Carter and Teramura 1988). Lianas are often characterized by their climbing type as roots, stem-twiners, or tendril climbers (Leicht-Young et al. 2010). Climbing mechanisms often dictates which tree hosts lianas can climb. Lianas with twining strategies require tree supports which are close together and small enough to curl around, while lianas with specialized adhesive structures can climb nearly any tree host large enough to support the liana’s weight (Ladwig and Meiners 2010).

Studies by Francis Putz (1984) suggest that lianas often fail to reach tree canopies or reproductive maturity because they are unable to locate appropriate structures to climb. A tree’s size (Putz 1984), bark texture, or production of allelopathic chemicals, can regulate the success of its climbing parasites (Ladwig and Meiners 2010). Only lianas with adventitious roots or adhesive tendrils are believed to have the capacity to climb large trees (excess of 30-40cm DBH) (Putz 1995). Observations by Putz (1984) proposed a strategy of relay ascension for vines which would otherwise be limited by their ability to only climb small trees. In northern Borneo and Washington, D.C., Putz (1984) observed vines that require small diameter trellises often climb large diameter trees by attaching to root and adhesive-tendril climbers (Putz 1994). Other studies have shown that larger tree hosts which already have a liana present are more likely to be colonized than smaller trees, or those without lianas (Ladwig and Meiners 2010). Relay ascension may have attributed to these findings.

This study investigated the prevalence of liana relay ascension at the E. S. George Reserve. Specific objectives of this study were to determine whether the presence of one vine on a host tree was indicative that many other vines would be found on that tree as well. Another goal was to determine whether a liana’s climbing mechanism dictates the order in which multiple lianas establish on host trees. The ultimate question in this study was whether certain vines climb other vines, as opposed to trees. The purpose of this study was to determine whether relay ascension is a common method for ascending large trees in temperate forests.
METHODS AND MATERIALS

This study was conducted at the E. S. George Reserve in Livingston County, Michigan, roughly 25km from Ann Arbor (42° 28' N, -84° 00' W). The reserve contains 525-hectares of private land owned by the University of Michigan. Fifty-three trees were sampled along the main Esker Road to collect data. Measures included the number of vines per host tree, the width and climbing method of primary, secondary, and final vines, and the distance between the primary and secondary vines. The occurrence of primary vine intersection and overlap was also recorded. The distance between primary and secondary vines was taken at the base of the tree. Intersection was counted whenever vines physically touched without directly overlaying each other. Overlap was counted whenever one vine specifically grew onto another vine as opposed to the host tree. Only trees with lianas present were included in this study.

RESULTS

Figure 1. The total number of vines per tree.

This graph shows the range in the number of vines that each host plant could support. Host trees had between 1-23 vines growing on them. The average number of vines per tree was 11.
Figure 2. The climbing mechanisms of primary, secondary, and final vines.

Figure 2 shows the number of vines with adventitious roots and adhesive tendrils that were classified as primary, secondary, or final. A Chi-Square test shows that the difference between the number of vines with adventitious roots or vines with adhesive tendrils in each category is significantly different (p=0.02). There are more lianas with adventitious roots which are primary vines than lianas with adhesive tendrils. There are more vines with adhesive tendrils than vines with adventitious roots which are final or secondary vines.

Figure 3. The percent occurrence of primary vine intersection.
Figure 3 shows the percent of vines within each climbing mechanism which experienced vine intersection. A Chi-Square test shows that the percentage of intersection of primary vines with adventitious roots was significantly greater than the percent of intersection for primary vines with adhesive tendrils (p=0.03).

Figure 4. The distance between primary and secondary vines.

Figure 4 shows the results of a linear regression between the number of vines on a host tree and the distance between the primary and secondary vines. There is a positive relationship between the total number of vines and the distance between the first two vines which is marginally significant (p=0.089). This graph suggests that vines on trees which host fewer vines grow closer together than on trees which host many vines. This trend did not explain very much of the variance observed (R² = 0.087).
Figure 5. The percent occurrence of primary vine overlap for vines with adventitious roots and adhesive tendrils.

Figure 5 shows the results of a Chi-Square test. I found that vines with adventitious roots are overlapped by other vines significantly more than vines with adhesive tendrils (p < 0.001).

Figure 6. Mean width of primary vines with adventitious roots and adhesive tendrils.
Figure 6 shows the results of a T-test. I found that primary vines with adventitious roots have a significantly greater width than primary vines with adhesive tendrils.

**DISCUSSION**

Results of this study support previous observations that trees with lianas already present are more likely to be colonized by other vines (Ladwig and Meiners 2010). Of the 53 trees surveyed in this study, only 3, (5%) supported only one vine. Solitary vines are unusual on the George Reserve, and predictably these trees are soon to acquire more vines. Only lianas with adventitious roots or adhesive tendrils were observed on the Reserve. An obvious trend showed that more primary vines have adventitious roots, while secondary and final vines have adhesive tendrils (Figure 2). Adventitious roots appear to ascend trees first, and vines with adhesive tendrils arrive on hosts afterwards. This pattern shows that there is an order to which lianas arrive on host trees and this order may be determined by climbing mechanisms.

Multiple vines on a single host tree often intersect. Results of this study show that primary vines with adventitious roots incur more vine intersection than primary vines with adhesive tendrils (Figure 3). Many vines intersected, seemingly haphazardly, but also many vines grew parallel to each other. Vines with adhesive tendrils tended to grow alongside each other without intersecting, while vines with adventitious roots were often crossed over by vines with adhesive tendrils. Beyond intersecting, vine overlap also occurs frequently. Primary vines with adventitious roots incur more vine overlap than primary vines with adhesive tendrils (Figure 5). The general trend observed was adhesive tendrils overlapped vines with adventitious roots. An obvious explanation for this pattern is the relative size of the adventitious roots to adhesive tendrils. The mean width of adventitious roots is nearly five times greater than the width of adhesive tendrils (Figure 6), thus it is easier for smaller vines to overlap large vines opposed to the alternative.

The colonization of a host tree by a liana with adventitious roots may result in more vines per tree than a host first colonized by a liana with adhesive tendrils. By overlapping adventitious roots, adhesive tendrils are able to execute relay ascension. Relay ascension may allow secondary vines to use larger vines for structural support as hosts instead of trees. Relay ascension of lianas may allow for greater biodiversity in forests because vines with different climbing mechanisms are able to climb both trees and other vines. This increase in biodiversity facilitated by relay ascension may also intensify the damage incurred by tree hosts as the weight of lianas increases.

Although the identification of species was limited in this study, based on previous knowledge of the lianas present on the George Reserve and their climbing mechanism, assumptions can be made as to which species were observed. Vines with adventitious roots were predictably *Toxicodendron radicans* or *Hedera helix*. Vines with adhesive tendrils may have been *Parthenocissus quinquefolia*, *Vitis* sp., or *Rubus* sp. (Burnham 2010). It is possible that the patterns of relay ascension observed in this study are species-specific. An interesting follow-up study might be conducted earlier in the season when leaves and fruits are present on the vines in order to determine which species are engaging in relay ascension.

Previous studies by Carter (1988) suggest that vines with adventitious roots, such as *Hedera helix* and *Toxicodendron radicans* may be restricted to growing only on large tree trunks and
large branches (Carter and Teramura 1988). Liana studies by Darwin in 1867 suggested that root-climbing mechanisms may be less efficient than twining strategies because adventitious roots require prolonged contact with a stable surface. Darwin predicted that this might prevent vines with adventitious roots from spreading onto the less stable, smaller branches in the highest portions of the canopy. Still, twining vines under the canopy may also be limited primarily by the reduced availability of smaller-diameter objects which provide a climbing path to full sunlight (Carter and Teramura 1988). On the other hand, Putz argued that only lianas with adventitious roots were able to climb large trees and reach canopy tops, suggesting that adventitious roots are more efficient (Putz 1995). The comparative success of climbing mechanisms in lianas is thus debatable, but the process of relay ascension may help to illuminate successful climbing strategies.

Lianas with tendrils may reach the path to full sunlight by first adhering to lianas with adventitious roots. As lianas with adventitious roots may be unable to exploit smaller branches, this would create resource partitioning on canopy tops. Lianas with adhesive tendrils may use relay ascension as a strategy to outcompete other species of lianas on the same tree. Lianas with adhesive tendrils are able to climb onto other larger vines whose growth is limited by the diameter of the tree, but then extend past vines with adventitious roots onto branches which are not yet colonized by vines. Utilizing relay ascension may provide a quick method to reaching canopy tops for smaller vines, creating a more successful strategy. The subject of relay ascension in lianas is not well known or documented. More studies should be conducted to better understand the mechanics of relay ascension, the interaction between species which climb, and the ways in which their collective climbing strategies affect forest ecosystems.

ACKNOWLEDGEMENTS
I want to thank Robyn Burnham for providing me with lianas keys, information on climbing mechanisms, and relay ascension. I want to thank Dave Allen for helping me to analyze the data, and my professors Ivette Perfecto and John Vandermeer for their support and encouragement.

WORKS CITED
Burnham, R. 2010. Censusing Lianas In Mesic Biomes of Eastern RegionS.


Abstract. A deciduous forest community in Livingston County, Michigan was investigated to comprehend the role an invasive shrub can have in the constantly changing understory of a primary forest. Japanese Barberry has been documented in the community since 1990 and I attempted to find any developed relationships or correlations occurring with the most abundant seedling species (Red Maple) that coexists with it in an 18 hectare plot known as the Big Woods. Densities of Red Maple seedlings underneath Japanese Barberry shrubs and the shrubs themselves were counted in ten 10 x 10m quadrats along with estimates of total surface area covered by barberry, canopy cover, and heights (cm) on an individual to individual basis. Plant height did not have a relationship with exposure to sunlight (P > 0.05), the mean height of Red Maples observed under Japanese Barberry was significantly taller than heights of randomly sampled Red Maples in the Big Woods plot (P < 0.05). There was no correlation with densities of Red Maple Seedlings and size classes of Japanese Barberry (P > 0.05) but a trend seems to be prevalent with the co-occurrence of Red Maple seedlings and plots with more Japanese Barberry.

INTRODUCTION

Invasive plant species can drastically alter the landscape of a community. The disruption of native fauna and flora dynamics can lead to cascading trophic effects. Often the direct relationship as to the various changes taking place will be fairly obvious. The indirect relationships evolved from the interaction of a native and invasive species and what consequence that may have for other native species are not as easily distinguished through other environmental noise (White, Wilson, Clark 2006). Furthering that difficulty is that many of the indirect relationships will not manifest themselves instantaneously and may take years beyond the invasive has established itself before the first hint of a correlation can be detected (Pearson 2009). The competitive edge that an invasive species has in the acquisition and utilization of resources typically biases research endeavors toward investigating possible negative scenarios and correlations (Daehler 2003). This case of negative consequences uniformly describes many direct relationships. However, it is a fairly recent trend that scientist have turned their efforts from the immediate consequences observed at the per individual basis (usually competition) and shifted towards understanding the integration of the invasive into the entire community structure.

The comprehensive study of how invasive species affect the community where they have been introduced has revealed that in some cases there emerge positive indirect relationships within the population, as a result of some species being disproportionately affected more than others. These relationships usually take longer to develop, but once they have been initiated can have long-standing implications for the future of the population. At the E. S. George Reserve in Livingston County, Michigan, the introduction of the Japanese Barberry (Berberis thunbergii) has been documented as early as the 1990s (Vandermeer, pers. comm.), although it is quite possible that it did inhabit parts of the reserve prior to this initial observation. The presence of
Barberry has had an alarming increase throughout the Reserve, and in time all indications are that it will dominate the landscape (Herron et al. 2007). The composition of the forest community where it has thus far emerged the most rapidly raises even more cause for concern. The Big Woods plot is host to a mostly Oak canopy and with a variety of species inhabiting the understory. Current estimates and censuses indicate that the succession of this current forest into a newly partitioned canopy is firmly underway, and the inclusion of a new invasive species has the capacity to alter the direction of current stages (Mosher, Silander and Latimer 2009). If the introduction of the Japanese Barberry increases competition for higher quality soils and light availability (which presumably the shrub will win), then the only species capable of withstanding the new environment would be shade tolerant individuals such as Red Maple (Acer rubrum), Witch Hazel, and several minor species in terms of abundance (Harrington, Fownes and Cassidy 2004). In order to extrapolate the possible ramifications of the invasive shrub in the Big Woods and throughout the George Reserve, I will be gathering baseline data about population densities and the spatial distribution of Red Maple Seedlings and Japanese Barberry shrubs in order to understand if there is an advantage to be had by shade tolerant species inherently by the possible shared preferences or ability to coexist with shrubs in the understory.

METHODS

The seedling and barberry survey was carried out on October 24th, 2010 on the E. S. George Reserve in Livingston County, Michigan. One hectare of forest on the E. S. George Reserve was chosen to investigate the relationships between Red Maple and Japanese Barberry. This hectare of forest located in the Big Woods plot was deemed suitable for the study for the occurrences of both Red Maple seedlings and Japanese Barberry shrubs throughout the hectare. From that hectare, one 100 x 10m plot was selected where the most frequent coexistence of the two species was found. The 100 x 10m plot was then broken up into ten 10 x 10m plots. Within each plot, every individual shrub of B. thunbergii was located and searched at its base for any A. rubrum seedlings. If present, the seedlings were measured for height along with the height of the shrub, and an estimate of canopy cover was made by densiometer on a per individual basis. For each plot, the total number of B. thunbergii shrubs, the percent coverage of surface area by barberry, and the total number of A. rubrum seedlings found under the branches at the base of each B. thunbergii shrub were counted.

All data analysis was completed in R.

RESULTS

Individual Seedling Comparisons

Height of individual seedlings did not strongly correlate with exposure to sunlight (Figure 1, R²= 0.001, P= 0.85). The gradient of canopy cover had a relatively broad range, and despite that there remained fluctuation in seedling heights, even to the extent of median height individuals being found in highly shaded areas.

Figure 1. The relationship between Red Maple seedling height and the percent canopy cover/ sunlight exposure
By contrast, the mean heights of *A. rubrum* growing beneath *B. thunbergii* were shown to be significantly larger (Figure 2, t= 2.9, d. f. = 34, P= 0.006) than Red Maple seedlings randomly selected throughout the Big Woods plots.

*Plot Level Comparisons*

The relative size of a barberry shrub showed a slightly inverted trend with densities of Red Maple seedlings present (Figure 3, $R^2 = 0.08$, P= 0.58).

Figure 2. The mean heights of randomly sampled Red Maple seedlings and those found beneath Japanese Barberry

Figure 3. Mean size of Japanese Barberry individuals and the densities of Red Maple Seedlings within that plot
In direct conflict with that notion, the number of individual Barberry shrubs seems to have a positive trend emerging when compared to densities of Red Maple seedlings found within the plot (Figure 4, $R^2 = 0.001$, $P = 0.45$). The relationship seems to suggest that more Red Maple seemed to occur where there was more Barberry, as long as the shrubs were not very large. This trend may reflect similar preferences or a positive association between the two species.

**DISCUSSION**

The positive correlation between densities of red maple seedlings found at the plot level and the densities of Japanese barberry best illustrates the underlying relationship between the two systems. When considered simultaneously with other factors which could explain the spatial
distribution of red maple seedlings, this particular relationship may be key to understanding the mechanism at work between the species and the implications it has for the future of the Big Woods plot and similar deciduous forests under invasion. Within the ten 10 x 10 m² quadrats the developing trend found with higher densities of red maple seedlings mirrored that of the higher densities of Japanese barberry shrubs. Total percentage of the quadrat area covered by *B. thunbergii* did appear to be in any form an indicator of where seedlings would have successful germination and growth. Although these two data seem counter-intuitive, the assumption can be drawn that the denser coverage of *B. thunbergii* may be limiting the growth rates and/or increasing mortality within seedlings sprouting beneath them. *B. thunbergii* shrubs reach a critical thickness at which they can completely shade out any undergrowth, with the additional function of preventing any canopy level leaf masses from accumulating on the soil, decreasing the nitrogen cycled back into the system surrounding the stems of *B. thunbergii*. The close association with high densities of barberry (individuals) within a plot and red maple seedlings suggests that seedlings have similar tolerances to abiotic factors. Of these conditions it would seem that canopy gaps are not as important, which was as expected due to the high shade tolerance of *A. rubrum* seedlings. In effect the relationship that may have developed between the two species may be one where because these two species seem to germinate in areas as a random function of seed dispersal and some undetected variable within the substrate they are choosing. This should give way to more or less to the observations made with *A. rubrum* seedlings often found with *B. thunbergii* of proportionally the same size.

Smaller barberry shrubs are not old enough to interfere substantially with the soil chemistry levels and in affect make them uninhabitable by seedlings. They do not have the physiology to completely shade out the smaller seedlings nor do they possess the ability to capture fallen leaf litter from the canopy cover. In time all of these characteristics of older shrubs will be realized, but *A. rubrum* may be tolerant of these conditions if it reaches a certain size class along with the shrub itself. Comparably similar size *A. rubrum* seedlings would have attributes that would leave them less vulnerable to the shading effects of the larger *B. thunbergii* because they would at least have some leaf surfaces exposed to sunlight if they have managed to grow up through some of the shrub. The disruptive growth through the branches of shrubs should give way to more “gaps” within the shrub and more openings for leaf litter to fall through, if the disruption itself does not drastically alter branching position of the shrub in a way that the leaf masses are able to reach the floor uncontested by the shrub. Finally, the root systems of both the shrub and the seedling are competing in the soils for nutrients and available resources. Similar sized seedlings as compared to shrubs would not be at as competitively a disadvantage, and even more importantly, the *A. rubrum* roots would be large enough that it is not confined to the highest concentrated areas of H⁺ ions in the soil layers at the base of *B. thunbergii* shrubs.

The presence of *B. thunbergii* at the E. S. George Reserve has projected exponential growth and will play a large part in the forest successions as we seem to be approaching a change in the community structure to many of the sub-canopy species that are partitioning themselves in the gaps with available light. The role of barberry in this shift seems to be indicated by the tolerance many of these sub-canopy species have to shade. If light availability is limiting for growth and survivorship more in some individuals than others, than for those species germinating in areas without barberry present will be of critical importance (Saccone 2010). This competition for what should presumably a very limited resource as *B. thunbergii* expands across the George Reserve should result in a very defined pattern of the spatial distribution of non-shade tolerant species clumped in canopy gap areas with no barberry. The other dimension
added to the process will be the possible release from deer browsing which holds many of the individual trees at browsing level until they have a season where they are untouched. With the barberry already possessing a deterrent attribute to deer browsing, any seedlings capable of growing amongst patches of *B. thunbergii* will have a huge advantage compared to the other species present. The compounded mechanisms of both confined spatial distributions and accelerated growth rates have substantial implications that the shade intolerant seedlings could be excluded from the forest community. The direct and indirect relationships between the *B. thunbergii* and the shade intolerant species at the George Reserve should serve as a model system for understanding the role that an invasive species can play in the successional stages of a forest community. The impact of that role and the composition of the forest species could be a clear indicator of the direction the succession of forest will take and ultimately which species transition into the canopy layer.

ACKNOWLEDGEMENTS

This study was made possible by the contributions of David Allen and John Guittar, without whom the locations of suitable forest conditions for both *B. thunbergii* and *A. rubrum* could not have been easily identified. Special thanks go to Debra Goldberg, Kate Zemenick, David Gonthier, Kassandra Semrau, John Guittar, John Vandermeer, Susan Cheng, Sahar Haghigat, Binbin Li, and Abigail Hyduke whose efforts contributed heavily to the preliminary findings associated with this study.

LITERATURE CITED


Diversity and Distributions. 12: 443-455.
PHENOTYPIC VARIATION OF *PRUNUS SEROTINA* BARK: IMPLICATIONS FOR THE INSECT COMMUNITY

KATELYN A. ZEMENICK

ABSTRACT

*Prunus serotina* Ehrhart (Rosaceae) is a common understory tree in the United States, and invasive in Europe. The bark of *P. serotina* is conspicuous, with rough irregular plates, comparable to “burnt potato chips”. *P. serotina* is known to have effects on the local insect community in its native range, and insects are commonly found in its bark plates. While variation in *P. serotina* bark is apparent, it has never been measured. This purpose of this study was to measure the variation in *P. serotina* bark and survey its effects on insect inhabitants – it may provide microhabitats for insect prey or predators, provide substrate for long term stay, or provide protection to substitute for parental care. It seems that the most important measure of variation is not the mean, but variation of bark chips for ovipositing Homopterans. There was a higher density of Homopteran egg cases on *P. serotina* individuals with greater variation in bark chips, suggesting that variation provides protection from predators and from the ambient environment.

INTRODUCTION

Black cherry, *Prunus serotina* Ehrhart (Rosaceae) is a common understory plant in the United States. In Michigan, *P. serotina* is common in disturbance-caused openings in dry-mesic and mesic forests, such as oak-hickory and beech-maple. It is also common in fence rows, old fields and edges of woodlots (Barnes and Wagner, 2004).

*P. serotina* has recently become of interest in Europe, where it has become a very successful invader since its introduction. Consequently, *P. serotina* has been studied extensively in Europe including its regeneration (Vanhellemont et al. 2009) ecophysiology (Phartyal et al. 2009) and factors affecting its radial growth (Vanhellemont et al. 2009). While much is studied to understand its control, there are few regarding its effects on native insect communities in Europe.

*P. serotina* is known to have effects on the local insect community in its native range. Tilman (1978) found that during the first three weeks of bud break, *P. serotina* has extrafloral nectaries which attract *Formica obscuripes* Forel, which are known predators of many insects found on *P. serotina*. Tilman found that *F. obscuripes*, during the time of budbreak, greatly reduced survivorship of *P. serotina*’s most common pest, the eastern tent caterpillar *Malacosoma americanum* (Tilman, 1978). In this case, *P. serotina* attracts predators of its own herbivore. While this is one example of *P. serotina*’s effect on the insect community, it is possible that there are many more.
When observed in the field, it is common to find spider webs in the bark of *P. serotina*, along with a suite of other insects. Beetles, ants and moths have all been observed underneath the bark plates, perhaps protecting them from the environment, from predators, or from being noticed by prey. The bark of *P. serotina* is conspicuous, with rough irregular plats, comparable to “burnt potato chips” and is therefore a key character for identification (Barnes and Wagner, 2004).

Previous studies have investigated the relationship of bark and insects. Most commonly studied are bark beetles, which are common pests in coniferous forests, which exploit the tree cambium for protection and food (Rudinsky, 1962). However, no studies were found that studied bark inhabitants that are not pests of the tree. This study seeks to understand if variation in *P. serotina* bark provide microhabitats for insect prey or predators, provide substrate for long term stay, or provide protection to substitute for parental care.

**MATERIALS & METHODS**

*Study Site*

The survey was performed on Sunday, October 17th, 2010 at the Edwin S. George Reserve. The reserve is located in Livingston County, Michigan.

*Survey Methods*

Eleven *P. serotina* individuals were surveyed in the Big Woods Plot, a 16-hectare permanent research plot in the ESGR. All individuals were within 37-56cm girth at breast height (GBH), and close in proximity. Six individuals were north of Esker Road, and 5 were south of Esker Road. At each individual, a string was tied at eye level, and other 10cm below eyeline. Within this area, the deviation of each bark ‘chip’ was measured. A chip was defined as a deviation from the trunk: a chip may constitute of one deviation or two, depending on if it was overlapping or if it was an exterior chip. The distance from the trunk was measured at the place of greatest deviation of the chip. The area was surveyed for live insects, spider webs, and egg cases.

*Statistical Analysis of Chip Size and Variation*

The frequency distribution of chip size for each *P. serotina* individual was fit with a lognormal distribution using the ‘plognorm’ function in R. The parameters in this distribution, the mean and the variance, were optimized using a maximum likelihood technique. The optimum parameter had the lowest value of negative log likelihood. A range of accepted values were used for the two parameters, the range being analogous to 95% confidence intervals. The range of values was used until the negative log likelihood of the model for the lowest negative log likelihood increased by two (Bolker, 2008). All comparisons of means were done with a resampling script in R, and general linear regressions were conducted in Microsoft Excel.
RESULTS

Of the 11 *P. serotina* individuals surveyed, 1,053 chips were measured, and 69 Homoptera egg cases were found. No live insects were present in any area surveyed.

**Mean Chip Size**

The mean chip size of *P. serotina* individuals was not different for trees N or S of Esker road (p>0.05). Within each site, there was no individual with a greater mean chip size than all of the rest. Further, there were no clear groupings of individuals with a greater mean chip size than another group (Table 1).

Table 1: Resampling Mean Chip Size for all combinations of *P. serotina* individuals surveyed. In each cell, the top number is the p-value, and the bottom is the mean difference. The matrix is set up so that the test would answer if the columns were greater than the row. Significant p-values are in bold.

<table>
<thead>
<tr>
<th></th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
<th>10</th>
<th>11</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.128</td>
<td>1.000</td>
<td>1.000</td>
<td>1.000</td>
<td>0.937</td>
<td>0.983</td>
<td>1.000</td>
<td>1.000</td>
<td>0.999</td>
<td>1.000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.031</td>
<td>-0.273</td>
<td>-0.202</td>
<td>-0.173</td>
<td>-0.043</td>
<td>-0.092</td>
<td>-0.164</td>
<td>-0.188</td>
<td>-0.081</td>
<td>-0.112</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>0.874</td>
<td>1.000</td>
<td>1.000</td>
<td>1.000</td>
<td>0.991</td>
<td>0.985</td>
<td>1.000</td>
<td>1.000</td>
<td>1.000</td>
<td>1.000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>-0.031</td>
<td>-0.304</td>
<td>-0.233</td>
<td>-0.203</td>
<td>-0.074</td>
<td>-0.123</td>
<td>-0.195</td>
<td>-0.218</td>
<td>-0.112</td>
<td>-0.143</td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>0.000</td>
<td>0.000</td>
<td>0.081</td>
<td>0.042</td>
<td>0.000</td>
<td>0.002</td>
<td>0.007</td>
<td>0.072</td>
<td>0.000</td>
<td>0.002</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.273</td>
<td>0.304</td>
<td>0.071</td>
<td>0.100</td>
<td>0.230</td>
<td>0.181</td>
<td>0.109</td>
<td>0.085</td>
<td>0.192</td>
<td>0.161</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>0.000</td>
<td>0.000</td>
<td>0.911</td>
<td>0.293</td>
<td>0.000</td>
<td>0.038</td>
<td>0.190</td>
<td>0.403</td>
<td>0.000</td>
<td>0.026</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.202</td>
<td>0.233</td>
<td>-0.071</td>
<td>0.030</td>
<td>0.159</td>
<td>0.110</td>
<td>0.039</td>
<td>0.015</td>
<td>0.122</td>
<td>0.090</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>0.000</td>
<td>0.000</td>
<td>0.958</td>
<td>0.704</td>
<td>0.000</td>
<td>0.121</td>
<td>0.431</td>
<td>0.596</td>
<td>0.012</td>
<td>0.137</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.173</td>
<td>0.203</td>
<td>-0.100</td>
<td>-0.030</td>
<td>0.129</td>
<td>0.080</td>
<td>0.009</td>
<td>-0.015</td>
<td>0.092</td>
<td>0.060</td>
<td></td>
</tr>
<tr>
<td>6</td>
<td>0.064</td>
<td>0.012</td>
<td>1.000</td>
<td>1.000</td>
<td>1.000</td>
<td>0.875</td>
<td>1.000</td>
<td>1.000</td>
<td>0.901</td>
<td>0.973</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.043</td>
<td>0.074</td>
<td>-0.230</td>
<td>-0.159</td>
<td>-0.129</td>
<td>-0.049</td>
<td>-0.121</td>
<td>-0.144</td>
<td>-0.038</td>
<td>-0.069</td>
<td></td>
</tr>
<tr>
<td>7</td>
<td>0.014</td>
<td>0.016</td>
<td>0.998</td>
<td>0.967</td>
<td>0.873</td>
<td>0.117</td>
<td>0.914</td>
<td>0.917</td>
<td>0.401</td>
<td>0.624</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.092</td>
<td>0.123</td>
<td>-0.181</td>
<td>-0.110</td>
<td>-0.080</td>
<td>0.049</td>
<td>-0.071</td>
<td>-0.095</td>
<td>0.012</td>
<td>-0.020</td>
<td></td>
</tr>
<tr>
<td>8</td>
<td>0.000</td>
<td>0.000</td>
<td>0.993</td>
<td>0.816</td>
<td>0.574</td>
<td>0.000</td>
<td>0.085</td>
<td>0.693</td>
<td>0.007</td>
<td>0.117</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.164</td>
<td>0.195</td>
<td>-0.109</td>
<td>-0.039</td>
<td>-0.009</td>
<td>0.121</td>
<td>0.071</td>
<td>-0.024</td>
<td>0.083</td>
<td>0.051</td>
<td></td>
</tr>
<tr>
<td>9</td>
<td>0.000</td>
<td>0.000</td>
<td>0.933</td>
<td>0.600</td>
<td>0.402</td>
<td>0.000</td>
<td>0.083</td>
<td>0.308</td>
<td>0.006</td>
<td>0.091</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.188</td>
<td>0.218</td>
<td>-0.085</td>
<td>-0.015</td>
<td>0.015</td>
<td>0.144</td>
<td>0.095</td>
<td>0.024</td>
<td>0.107</td>
<td>0.075</td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>0.002</td>
<td>0.000</td>
<td>1.000</td>
<td>1.000</td>
<td>0.987</td>
<td>0.097</td>
<td>0.608</td>
<td>0.992</td>
<td>0.994</td>
<td>0.994</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.081</td>
<td>0.112</td>
<td>-0.192</td>
<td>-0.122</td>
<td>-0.092</td>
<td>0.038</td>
<td>-0.012</td>
<td>-0.083</td>
<td>-0.107</td>
<td>-0.107</td>
<td></td>
</tr>
<tr>
<td>11</td>
<td>0.001</td>
<td>0.000</td>
<td>0.999</td>
<td>0.971</td>
<td>0.861</td>
<td>0.023</td>
<td>0.371</td>
<td>0.887</td>
<td>0.906</td>
<td>0.190</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.112</td>
<td>0.143</td>
<td>-0.161</td>
<td>-0.090</td>
<td>-0.060</td>
<td>0.069</td>
<td>0.020</td>
<td>-0.051</td>
<td>-0.075</td>
<td>0.032</td>
<td></td>
</tr>
</tbody>
</table>

The number of chips measured on each individual tends to be related with greater mean chip size, but this relation was not significant. Larger trees tend to have more chips (p=0.184, R^2= 0.187) and a larger mean chip size (p= 0.011, R^2=0.533). However, the number of chips was not correlated with mean chip size (p=0.556, R^2=0.038).
Variation in Chip Size

Like mean chip size, the variation in chip size was not different for individuals N or S of Esker Road. The variation of chip size was not affected by the size of the tree \((p = 0.699, R^2 = 0.017)\). Unlike mean chip size, there were distinct groups of individuals that had a low, medium or high variation in chip size (Figure 1). Based on their negative log likelihood, the first three individuals are significantly different from the last three individuals seen on the figure. Again, these differences were not attributable to the location or size of the tree.

![Figure 1: Variation of Chip Size in P. serotina](image)

Density of Homopteran Egg Cases

Homopteran egg cases were abundant on \(P. serotina\) individuals. They contained 4-6 crawlers covered by two layers of silk webbing (Figure 2). The density of egg cases on \(P. serotina\) individuals ranged from 0 to 0.032 case/cm\(^2\). A weak trend suggests that more mats are present on larger trees \((p = 0.358, R^2 = 0.094)\). The egg cases seemed to be clumped in space, but the density of egg cases was not correlated with tree size \((p = 0.482, R^2 = 0.056)\). There was no relationship between mean chip size and the density of egg cases \((p=0.503, R^2=0.051)\). However, there was a strong trend indicating that a greater density of egg cases were present on \(P. serotina\) individuals with a greater variance in chip size (Figure 3).
DISCUSSION

Of *P. serotina* individuals close in proximity and size, and seemingly in age, there are differences in the variation of chip size. Further, as the variance of chip size increases, the density of Homopteran egg cases increases. However, the density of egg cases does not change relative to mean chip size. This indicates that the placement of egg cases is not dependent on the mean chip size, but the variation of the chip size. *P. serotina* bark does seem to be of importance to egg case location.

Assuming the egg cases are not placed haphazardly, it is interesting to question why they are placed where. In this case, it seems that the Homopteran adults are placing more egg cases on *P. serotina* individuals with greater chip variance. Larger chips may protect the
egg cases, it was observed that some egg cases were placed on the underside of chips, or on a chip that was partially covered up by another chip. Therefore, larger chips may actually protect the egg cases from predators or from the environment. However, larger chip sizes may provide a refuge for other insects. It is possible that these ‘other’ insects may predate upon the Homopterans found in the egg cases. Hence, larger chips may both protect the Homopterans or house their predators. Adults placing the egg cases may be trying to weigh these costs and benefits, and the choice of bark with variable chip size may optimize these tradeoffs. Therefore, the apparent placement of the egg cases could be a form of parental care.

Of course, it would be interesting to survey other tree species present in ESGR to determine if these Homopterans selectively inhabit *P. serotina*, tree species with chippy or flakey bark, or if they have no species preference at all.

It would be interesting to understand the cause of the phenotypic variation seen in the 11 *P. serotina* individuals surveyed. As there was not a difference of mean chip size or variance of chip size in the two sites, it might seem plausible to say that the phenotypic variation is not attributed to plasticity. However, there is little basis for this claim, and plasticity may be related to environment during different ontogenic stages.

Whether or not plasticity is important, it is likely that genetic variation is definitely important in chip size or variance in *P. serotina*. Pairon et al. (2010) found that *P. serotina* originate from eastern Appalachia, and a key to its invasiveness in Europe is due to multiple introductions of large population sizes. This explanation may pertain to the ESGR – *P. serotina* success may be due to genetic variability, and the differences in chip size observed may therefore be due to genetic variation.

ACKNOWLEDGMENTS

I would like to thank Dave Allen for his aid in statistical analysis, and John Vandermeer and Ivette Perfecto for the help in identifying and photographing the Homopteran crawlers.

WORKS CITED


