

University of Memphis

University of Memphis Digital Commons

---

Electronic Theses and Dissertations

---

11-28-2011

## The Effects of Habitat Fragmentation on Arthropod Biodiversity in Native Canebrakes

Justin James Geise

Follow this and additional works at: <https://digitalcommons.memphis.edu/etd>

---

### Recommended Citation

Geise, Justin James, "The Effects of Habitat Fragmentation on Arthropod Biodiversity in Native Canebrakes" (2011). *Electronic Theses and Dissertations*. 370.

<https://digitalcommons.memphis.edu/etd/370>

This Thesis is brought to you for free and open access by University of Memphis Digital Commons. It has been accepted for inclusion in Electronic Theses and Dissertations by an authorized administrator of University of Memphis Digital Commons. For more information, please contact [khhgerty@memphis.edu](mailto:khhgerty@memphis.edu).

THE EFFECTS OF HABITAT FRAGMENTATION ON ARTHROPOD  
BIODIVERSITY IN NATIVE CANEBRAKES

by

Justin James Geise

A Thesis Submitted

in Partial Fulfillment of the

Requirements for the Degree of

Master of Science

Major: Biology

The University of Memphis

December 2011

## **Abstract**

Geise, Justin J. M.S. The University of Memphis. December 2011. The Effects of Habitat Fragmentation on Arthropod Biodiversity in Native Canebrakes. Major Professor. S. Reza Pezeshki, Ph.D.

Dense stands of *Arundinaria gigantea* (Walt.) Chapm. or canebrakes were common in the southeastern US, but currently are a critically endangered ecosystem with less than 2% of their area remaining. Remnant canebrakes have become fragmented and isolated. In this study, arthropod communities within these canebrake patches and non-cane patches were examined to assess general island biogeography theories. Four patches (>0.5ha; <0.5ha to >0.1ha; <0.1ha; adjacent non cane patch) were selected at four sites in the Mississippi alluvial valley during three collection periods. Measures of diversity, community composition, and functional group organization were compared to determine if differences existed. Diversity measures and functional group organization did not differ among patch sizes. Diversity measures decreased as distance from canebrakes increased. Canebrakes contained many species that were not found in adjacent non-cane patches with almost 97% uncommon to rare. Conservation of canebrakes is necessary to maintain the biodiversity of these uncommon arthropods.

## **Preface**

The main body of this thesis consists of a manuscript written in a style appropriate for the journal Environmental Entomology. Appendices have been added to the thesis in order to provide supporting data for interested investigators.

## Table of Contents

	<b>page</b>
List of Figures	iv
List of Tables	v
Introduction	<b>1</b>
Objectives and Hypotheses	<b>5</b>
Materials and Methods	<b>5</b>
Results	<b>9</b>
Discussion	<b>15</b>
Literature Cited	<b>19</b>
Appendix	<b>51</b>

## List of Figures

	page
<b>Figure 1.</b> a) Total abundance of arthropods at each site and collection period. b) Total evenness at each site and collection period.	32
<b>Figure 2.</b> . Total Shannon-Wiener diversity ( $H'$ ) at each site and collection period.	33
<b>Figure 3.</b> Linear regression of the relationship between a) evenness and b) abundance at various canebrake patches.	34
<b>Figure 4.</b> Linear regression of the relationship between a) species richness and b) Shannon-Wiener Diversity at various canebrake patches	35
<b>Figure 5.</b> Linear regression of the relationship between a) species richness and b) abundance with the distance to nearest canebrake in all cane patches.	36
<b>Figure 6.</b> Linear regression of the relationship between a) evenness and b) Shannon-Wiener Diversity and distance to nearest canebrake in all cane patches.	37
<b>Figure 7.</b> a) A comparison of functional group organization at each collection period. b) A comparison of functional group organization all patch sizes at all site and all collection periods.	38
<b>Figure 8.</b> a) Linear Regression of the relationship between Shannon-Wiener Diversity with distance to nearest patch in a) large cane patches and b) intermediate cane patches.	39
<b>Figure 9.</b> a) Linear Regression of the relationship between Shannon-Wiener Diversity with distance to nearest patch in small cane patches.	40

## List of Tables

	<b>page</b>
<b>Table 1 .</b> Canebrake patch area and distance to nearest patch at each study site.	<b>24</b>
<b>Table 2.</b> Collection dates for each collection period at each site.	<b>24</b>
<b>Table 3.</b> GLM (Type III Sums of Squares) and GLM MANOVA results of arthropod community response variables.	<b>25</b>
<b>Table 4.</b> Results of MRPP analysis of patch size and collection and indicator species Analysis of functional groups in native canebrakes and adjacent bottomland hardwood forest.	<b>25</b>
<b>Table 5.</b> Indicator species analysis for the 20 most abundant species found at each site.	<b>26</b>
<b>Table 6.</b> Community similarity for all patches at each collection period at each site.	<b>27</b>
<b>Table 7.</b> Repeated measures ANOVA and MANOVA results of arthropod community response variables.	<b>31</b>
<b>Table 8.</b> Species occurrence and abundance at canebrake patches and adjacent bottomland hardwood forest at all four study sites.	<b>43</b>

## Introduction

*Arundinaria gigantea* (Walt.) Chapm. or giant cane is a woody grass (Poaceae) found in the southeastern US. Giant cane is most commonly found on stream banks, terraces, and as a midstory component in bottomland and mesic hardwood forests (Campbell 1985). Giant cane reproduces sexually and asexually. This asexual reproduction of giant cane can form dense, monotypic stands which are also known as canebrakes. Giant cane is a semelparous species and has a life expectancy of around 25 years. Giant cane is also a fire and flood tolerant species; both were historically common in the southeastern US (Hughes 1966).

Historically, dense stands of giant cane (i.e., canebrakes) were a common feature of the landscape in the Southeastern United States. In 1908, Theodore Roosevelt wrote, “canebrakes extended for miles” and “canebrakes are refuges of hunted things.” Some of these canebrakes were “well-nigh impenetrable to a man on horseback.” In 1818, John James Audubon and Constantine Rafinesque were exploring a canebrake near Audubon’s cabin in Kentucky and the two got lost in the canebrake and were nearly attacked by a black bear, subsequently ending Rafinesque’s adventures in canebrakes (Kastner 2000). It is speculated that canebrakes once occupied over 120,000 ha in just Alabama and Kentucky (Campbell 1985; Barrone et al. 2008). Today, canebrakes occupy <2% of their original area because of land use changes, floodplain hydrology alterations, and fire suppression (Noss et al 1995; Platt and Brantley 1997; Stewart 2007). As a result, remnant canebrake stands are isolated fragments.



Fragmentation is believed to be one of the major causes of the decline of biodiversity worldwide (Butchart et al 2010). Fragmentation of continuous habitat into smaller, more isolated areas results in a loss of species due to a reduction in remnant area (Didham et al. 1996). This reduction leads to an increase in isolation and edge and ultimately a decrease in connectivity (Harrison and Bruna 1999). The theory of island biogeography has often been used to explain the loss of diversity (measured as abundance, richness, or diversity) as a function of habitat area loss resulting from fragmentation (Andren 1994). Predictions supporting mainland versus island dynamics suggest that mainland richness will be greater than islands of equal size quadrats (Preston 1962). Remnant canebreaks are much different from oceanic islands because they are surrounded by multiple anthropogenic habitats (i.e., matrix) and, therefore, subjected to greater anthropogenic disturbances, such as agriculture, fire suppression, and altered hydrological patterns (Andren 1994). This surrounding matrix, particularly in terrestrial habitats, is not always inhospitable, is capable of supporting populations, and can provide corridors of dispersal between patches (Jules and Shanani 2003; Kupfer et al 2006).

Fragmentation effects on arthropod biodiversity have been a topic of great interest to scientists. However, results have been contradictory. The Equilibrium Theory of Island Biogeography states that larger, less isolated areas have higher biodiversity and less extinction events than smaller, more isolated areas (MacArthur and Wilson 1967). The Equilibrium Theory of Island Biogeography can be applied to terrestrial habitats if the “islands” are defined correctly from the viewpoint of the target species (Lovei et al. 2006). Much of these contradictory results (Krauss et al 2003; Martinko et al 2006) stem from failing to account for matrix dwelling species, crowding effects in remnant

fragments, or even a lack of understanding of the life history of the organisms studied (Debinski and Holt 2000; Ewers and Didham 2006). Many studies have shown that fragmentation negatively affects insect biodiversity and found higher insect diversity in larger patches compared to smaller patches (Karieva 1987; Bierregaard et al 1992; Miyashita et al 1998; Collinge 2000). Didham et al (1998) showed that common beetle species declined by 49.8% when tropical forest fragments were decreased to 1 ha. In another study, butterfly specialist and generalist species became rarer and specialist extinction rates rose as fragmentation increased over 100 years in calcareous grasslands in Belgium (Polus et al 2007). Fragmentation has also been shown to reduce diversity of pollinator communities and insect pollinated plant species (Taki et al. 2007).

Fragmentation may affect trophic processes and alter the organization of trophic levels with a decrease in parasitoids and an increase in generalist predators. (Valladares et al. 2006). Layman et al (2007) stated that fragmentation can destabilize food webs, which in turn causes a collapse in trophic levels. Understanding of the effects of fragmentation on biodiversity is timely due to quickly changing land use and destruction of habitat.

Canebrakes provide unique and important ecosystem services. There are 50 species of animals documented in canebrakes: 23 mammals, 16 birds, four reptiles, and seven species of invertebrates (Platt et al. 2001). There are at least seven species that are cane obligates: six are butterflies (Lepidoptera) and one bird, Bachman's warbler (*Vermivora bachmani*) (Brantley and Platt 2001). Bachman's warbler is an obligate of native canebrakes and the rarest songbird in North America and it is presumed extinct considering that the last official sighting was in 1962 (Hamel and Hooper 1979; Remsen 1986). It is speculated that the Bachman's warbler's fate is due to the loss of continuous

native canebrakes (Remsen 1986) having used canebrakes for nesting habitat and arthropod foraging (Remsen 1986). Bachman's warbler primarily fed on ground-dwelling arthropods such as Carabidae (ground beetles), Formicidae (ants), Gryllidae (crickets), and Araneae (spiders) (Meanly 1966). Five of the six butterfly species obligately associated with canebrakes are listed as species of concern due to habitat destruction (Opler 1992; Platt et al 2001). The Louisiana black bear is another species that uses canebrakes and is also threatened (Platt et al 2001). Passenger pigeons, once the most abundant bird in North America and possibly on Earth before becoming extinct in 1914, were often found roosting in canebrakes (Schorger 1955; Platt et al 2001). Canebrakes in the Southeastern United States are an important ecosystem due to the ecological necessity of certain wildlife and invertebrate species. Determining the relationship between arthropod diversity and fragmented canebrakes will give us insight on how landscape fragmentation affects biodiversity of arthropods.

Current research on canebrakes involves studies of reestablishment for stream bank stabilization (Schoonover and Williard 2003; Dattilo and Rhoades 2005; Baldwin et al. 2009) and the ecology and physiology of canebrakes (Cirtain et al 2004; Gagnon 2009). To date, however, arthropod communities within canebrakes have received very little attention. There are no known studies that have documented the species composition of arthropod communities in canebrakes with the exception of butterflies (Platt et al. 2001). In order to understand trophic structure in canebrakes, arthropod communities and their associations need to be examined.

## **Objectives and Hypotheses**

To address this topic, a natural study was designed to characterize and compare the arthropod communities in both canebrake and non-canebrake patches of varying sizes to assess general theories of biogeography and to determine if specific arthropod communities exist in canebrakes. It was hypothesized that larger patches of giant cane would have higher arthropod diversity when compared to smaller patches and that community composition and similarity would differ among patch sizes. Secondly, it was hypothesized that more isolated patches (i.e., greater distances between patches) will have less diversity than less isolated patches (i.e., distance between patches). This isolation should reduce community similarity. The third hypothesis was that functional group dominance will vary by patch size. The last hypothesis was arthropod communities had higher temporal fluctuations in abundance, species richness, Shannon-Weiner Diversity, evenness, and functional group organization in smaller patches compared to larger patches.

## **Materials and Methods**

**Study Sites.** Four study areas were selected due to the presence of varying canebrake patch sizes; two in Tennessee, one in Arkansas, and one in Mississippi. In Tennessee, one area was located in Meeman-Shelby State Park in northwest Shelby County (35.3789° N, -90.0269° W) and the other was W.C. Johnson Park in southeastern Shelby County (35.0894°, -89.6689°). The Arkansas site was located in Cache River National Wildlife Refuge in eastern Prairie County (34.7963°, -91.3617°). The Mississippi site was located in Dahomey National Wildlife Refuge in western Bolivar County (33.6883°,

-90.9015°). All four study areas are situated in a humid, subtropical region of the Northern Temperate Zone. Annual precipitation ranges from 1000 mm to 1600 mm and average temperatures range from 11°C in January to 27°C in July (Gardinier and Oliver 2004). Soils within the study area consist of alluvial silts and clays with a high content of organic matter (personal observation). All four study areas were bottomland hardwood forests primarily composed of species such as oak (*Quercus nigra* L.); (*Quercus pagoda* Raf. ), hickory (*Carya tomentosa* (Poir.) Nutt.); (*Carya glabra* (Mill.)), elm (*Ulmus Americana* L.) ,and hackberry (*Celtis occidentalis* L.) which is common in the Mississippi Alluvial Valley (personal observation). All canebrakes in the study areas exist as the midstory component of these forests. All canebrakes with the exception of the Collierville site are proximate to agricultural lands. The Collierville site is proximate to a residential subdivision.

**Procedures.** At each site, four patch size classes (chosen based on canebrake fragments found at each site) were sampled: 1) canebrakes <0.1 ha, 2) canebrakes between >0.1 ha and < 0.5 ha, 3) canebrakes >0.5 ha., and 4) a non canebrake area adjacent to the other size classes to examine matrix use by arthropods and to define a canebrake community (Table 1). At each of the four sites per patch size class, two sampling methods were employed. First, 20 pitfall traps were installed along two 50 m transects for intermediate, large, and non-cane areas and 10 m transects for the small patch areas, with ten pitfall traps along each transect set equidistant apart. The pitfall traps were 1m apart for the smallest patches (due to limited area) and 5m apart for the intermediate, large, and non cane patches. Ward et al (2001) stated that smaller spacing intervals reduce pitfall trap

efficiency, but for this study, the smaller spacing interval was necessary in order to fit within the smallest canebrake patches. Jars (0.5 L) were used for each pitfall trap and were open for one 24-hour period on three occasions during the growing season (from late May to October 2008) (Table 2). Collection periods were separated into three time periods (May 29-June 26 2008; July 6-August 6 2008; August 15-October 20 2008). The second method consisted of manually collecting insects from 20 randomly selected culms of giant cane at each site. After each collection period, all arthropods were placed in 70% ethanol. All canebrake patches were physically measured and GPS coordinates and GIS software were utilized to determine patch area and distance to nearest canebrake patch (Table 1), a measure to determine how isolated cane patches were from each other.

**Analysis.** Arthropods were identified to the lowest possible taxonomic level and were categorized into functional groups. Four different measures of arthropod communities were calculated by both taxa and functional group: abundance, species richness, Shannon-Weiner diversity ( $H'$ ), and evenness ( $E_H$ ). Abundance was measured as the total number of individuals collected. Species richness was measured as the total count of the number of different species collected. Shannon-Weiner diversity was determined by using the formula  $H' = -\sum P_i(\ln P_i)$ , where  $P_i$  is the proportion of the abundance of a particular species with the total amount collected. Evenness was calculated as  $E_H = H' / \ln(\text{total abundance})$ .

The first hypothesis was that larger cane patches had higher community diversity measures and differed in community composition compared to smaller patches. In order to test this hypothesis, a multiple factor analysis of variance (MANOVA) was undertaken to determine if community measures (evenness, species richness, abundance, and

Shannon-Wiener diversity) differed by patch size, site, and collection period. Individual GLM's were subsequently performed on statistically significant factors in the MANOVA. Tukey's post hoc comparisons were used to determine pair-wise differences. A Multi Response Permutation Procedure (MRPP) was performed on all species at each patch size and collection period. A MRPP measures the composition differences among groups by calculating average similarity within groups and comparing those to among groups (a delta statistic). The P value is the proportion of smaller deltas and tests the hypothesis that there are no differences in community composition among groups. A linear regression was used to determine if any relationship existed between community measures with canebrake patch area. Percent community similarity (abundance values) was also calculated for each patch size, site, and collection period as a quantitative measure of community variability over time. Community similarity was used to determine how Arthropod communities differed by patch size and collection period. Community similarity compares how similar composition of one patch is to another. This was accomplished by the equation  $\sum (\min (p_{\text{patch a}} p_{\text{patch b}})) * 100$ ; P is the proportion of the abundance of a particular species with the total amount collected. An Indicator Species Analysis (ISA) was used to determine if particular species within the community favored a particular patch size. ISA is an analysis that weighs maximum value between groups with specificity (individuals and their relative abundance) and site fidelity which is measured by indicator value (IV).

The second hypothesis was that the more isolated patches have lower measures of diversity than less isolated patches. Linear regression was used to address the second

hypothesis, determining the relationship between diversity measures and distance to nearest canebrake patch.

Arthropods were separated into functional groups and were analyzed by determining the proportion of each group (predators, omnivores, herbivores, scavengers, and parasites) by the total amount collected at each site, patch size, and collection period. The proportion of each functional group was arcsine transformed prior to analysis. The third hypothesis was that smaller cane patch functional group organization would differ from larger patches.

To test the third hypothesis, a perMANOVA was used to determine if functional group abundances were different at each patch size. An indicator species analysis was used to determine if functional groups differed by patch size. The last hypothesis was addressed using Repeated measure ANOVA's to determine if community diversity measures differed among patch sizes and collection periods (collection period for the Repeated measures ANOVA is evaluated at each collection point at each patch size and site). The coefficients of variation from the repeated measure ANOVA's were used to determine if variability among patch sizes and collection period increased or decreased temporally.

## **Results**

**Species Collected.** Approximately 400 different taxa were found in all pitfall traps and by manual picking. The most common taxa were ants (Hymenoptera:Formicidae), ground beetles ( Coleoptera: Carabidae), mites (Order Actinedida), and springtails (Colembola: Entobryionidae: Isotomidae; Sminthuridae). These most common taxa were



found in all patch sizes. Occurrences of different taxa ranged from ubiquitous to rare: 53.3% of all species collected in pitfall traps were only found in canebrakes and 96.9% of these species were uncommon to rare (<10 individuals collected). Only 8.7% of the species collected were found in non cane patches: all of these species were uncommon to rare (<10 individuals collected). Due to the very small proportion of these rare to uncommon insects in the community, they were not detected through MANOVA results

**Community Measures Analysis.** MANOVA results show that there was a significant difference in community measures (abundance, evenness, species richness, and Shannon-Wiener Diversity) by collection period and site\*collection period (Table 3). Community measures were not different among patch sizes. The difference in collection periods indicated that there was a temporal shift in community measures during the growing season. The interaction between patch size and collection period was not significant (Table 3), suggesting that community measures were different due to the variation in time period regardless of patch size.

**Community Composition.** MRPP results showed that community composition did not differ by patch size, but composition was different by collection period (Table 4). Indicator Species Analysis found that of the 20 most abundant species found, eight had higher relative abundance in intermediate cane patches (Table 5). Community similarity was usually higher among patches collected at the same time period compared to patches of similar size within different collection periods (Table 6).

**Abundance.** Repeated measures ANOVA detected differences in abundance at site (Repeated Measures ANOVA,  $F=8.57$ ;  $p>F_{3,18}=0.0053$ ), collection period (Repeated Measures ANOVA,  $F=7.99$ ;  $p>F_{2,18}=0.0033$ ), and site \* collection period (Repeated Measures ANOVA,  $F=3.46$ ;  $p>F_{6,18}=0.0189$ ; Table 7). This indicates that there was a change in abundance at sites over the three collection periods, but this change was not the same for all sites. Abundance was significantly different by site (ANOVA,  $F=4.93$ ;  $p>F_{3,18}=0.0113$ ), collection period (ANOVA,  $F=7.99$ ;  $p>F_{2,18}=0.0033$ ), and site\* collection period (ANOVA,  $F=3.46$ ;  $p>F_{6,18}=0.0189$ )( Table 5). Abundance data showed the least variability during the first collection period and the highest variability during the second collection period (Abundance CV: T1 = 31.0; T2 = 44.1; T3 = 38.7). Abundance was higher at the Collierville site and significantly different from the Cache River and River Road sites (Figure 1a). The first collection period had significantly lower abundance compared to collection periods two and three (Figure 1a).

**Evenness.** Evenness was significantly different due to site (Repeated Measures ANOVA,  $F=5.12$ ;  $p>F_{3,18}=0.0245$ ; Table 7), collection period (Repeated Measures ANOVA,  $F=4.05$ ;  $p>F_{3,18}=0.0354$ ; Table 7), and the collection period \* site interaction (Repeated Measures ANOVA,  $F=8.57$ ;  $p>F_{6,18}=0.0053$ ; Table 7; Figure 1b). This indicates that there is a change in evenness over collection periods but the change was different among sites. Collierville showed a greater decrease in evenness as the growing season progressed compared to other sites (Figure 1b). Evenness also significantly differed among collection periods (ANOVA,  $F=4.05$ ;  $p>F_{3,18}=0.0354$ ; Table 3): collection period one had significantly higher evenness than collection periods two and three (Figure 2b).

Cache River had higher evenness at collection period three while Collierville had the least. Coefficient of Variation (CV) showed that variability of evenness was least during the last collection period (Evenness CV: T1 = 14.7; T2 = 16.8; T3 = 11.9)

**Species Richness.** Repeated measures ANOVA and GLM ANOVA results showed that species richness was not significantly different by patch size, site, or collection period (Table 3, 7). Species richness showed a significant interaction between site and collection period (Table 3, 7). Coefficient of Variation (CV) for diversity measures showed that variability for species richness was least during the last collection period (Species Richness CV: T1 = 19.7; T2 = 19.5; T3 = 18.9).

**Shannon-Wiener Diversity.** Repeated measures ANOVA show that  $H'$  had a significant collection period \* site interaction (Repeated Measures ANOVA,  $F=3.60$ ;  $p>F_{6,18}=0.0161$ ; Table 7). This indicates that there was a change in ( $H'$ ) at sites over the three collection periods, but this change was not the same for all sites. Cache River had higher ( $H'$ ) at collection period one and three compared to collection period two (Fig 7a). Collierville had higher ( $H'$ ) at collection period one compared to collection periods two and three (Fig. 7a). River Road and Dahomey were not different between collection periods (Fig 7a). Coefficient of Variation (CV) for diversity measures showed that variability was least during the last collection period (Diversity CV: T1 = 12.05; T2 = 14.9; T3 = 8.24)

**Species-Area Relationships in Canebrakes.** Canebrake area was not statistically significant when regressed with evenness ( $R^2=0.27$ ,  $F_{1,34}=0.946$ ,  $p=0.338$ ; Fig. 3a), abundance ( $R^2=0.03$ ,  $F_{1,34}=0.099$ ,  $p=0.755$ ; Fig 3b), species richness ( $R^2=0.06$ ,  $F_{1,34}=0.216$ ,  $p=0.645$ ; Fig.4a), or Shannon-Weiner Diversity ( $R^2=0.17$ ,  $F_{1,34}=0.591$ ,  $p=0.447$ ; Fig. 4b). The results further suggest that there was not a difference in community diversity as patch size of canebrakes increased.

**Nearest Canebrake Relationships.** Distance to nearest canebrake was statistically significant when regressed with Shannon-Weiner Diversity ( $R^2=0.113$ ,  $F=4.349$ ,  $p=0.045$ ; Fig. 6b). Shannon-Wiener Diversity had a strong negative relationship when regressed on distance to nearest canebrake in large cane patches ( $R^2=0.962$ ,  $F=51.208$ ,  $p=0.019$ ; Fig.8a) and intermediate cane patches ( $R^2=0.863$ ,  $F=12.563$ ,  $p=0.071$ ; Fig. 8b). Distance to nearest canebrake in small cane patches ( $R^2=0.166$ ,  $F=0.398$ ,  $p=0.593$ ; Fig. 9a) was not significant. Distance to nearest canebrake was not statistically significant when regressed with evenness ( $R^2=0.098$ ,  $F=3.695$ ,  $p=0.063$ ; Fig. 6a), abundance ( $R^2=0.046$ ,  $F=1.621$ ,  $p=0.212$ ; Fig. 5b), or species richness ( $R^2=0.008$ ,  $F=0.264$ ,  $p=0.611$ ; Fig. 5a). Abundance had a positive relationship with distance to nearest canebrake which may indicate a species crowding effect in these isolated canebrakes. Isolation did not drastically influence community similarity; the Collierville intermediate cane site was the most isolated (Table 1) and had comparable community similarity (Table 6) to sites that were less isolated.

**Functional Group Organization.** Functional group composition did not differ by patch size but was significantly different by collection period (perMANOVA,  $F_{2,45}=7.9293$ ;  $p=0.0002$ ). Pair-wise comparisons indicated that functional groups were different at each collection period. Scavengers were significantly higher at later collection periods while phytophagus and omnivores taxa decreased as the growing season progressed (Figure 7a). Indicator species analysis also detected a difference for scavengers during different collection periods (ISA,  $IV=58$ ,  $p=0.0004$ ; Table 4)

## Discussion

Fragmentation of the landscape is one cause of worldwide biodiversity decline (Butchart et al. 2010). Important ecological literature such as McArthur and Wilson (1967) state that as area increases, diversity increases. However, in this study, area and patch size were not a factor for measures such as species richness, Shannon-Weiner Diversity, abundance, and evenness. These results were confounding but not uncommon. Factors such as species life history, matrix effects, crowding effects, and edge effects have shown that area is not always the main factor in determining species diversity (Ewers and Didham 2006). Habitat quality may be as important in determining diversity as patch size and isolation (Dennis and Eales 1997). Unfortunately, habitat variables beyond isolation and area were not measured in this study.

We hypothesized that species diversity would increase in larger canebrake patches and community composition would be different among patch sizes. The results suggest that species diversity measures were not influenced by canebrake patch size. Studies have shown that area is not always a predictor in arthropod diversity (Didham et al 1996). Patch size appeared not to be a determinant for diversity measures; however, canebrakes contained 196 species (or 54% of the total number of species) that were not found in adjacent non cane patches. Of these, 97% were uncommon to rare species (<10 total individuals collected). This indicates that canebrakes may contain a unique Arthropod community that is not found in adjacent forest. It is not known if these species select for cane habitat or if these species occur by chance.

Since the majority of canebrakes have been lost over the last century, it may be safe to assume that all cane patches in this study could be considered small fragments in comparison to the large continuous canebrakes found historically. Small fragments have been shown to have different and unique species assemblages compared to larger continuous patches (Gibb and Hochuli 2002; Tschardt et al 2002). The unique species assemblage in small fragments may be caused by pressures such as anthropogenic alterations, colonization of new invaders, and release from natural enemies (Yahner 1988), but the present data showed no evidence for differences in assemblages based on patch size.

Fragmentation can alter trophic level interactions due to alterations in abundance and occurrence of key species, which in turn can destabilize food webs (Laurence 2002; Terborgh et al 2001; Layman et al 2007). Typically, parasitoids and specialists decrease in highly fragmented landscapes and edges while predators (generalist) increase in edges (Valladares et al 2006). In this study, functional group organization was not statistically significant in relation to patch size. Arthropod functional group organization did differ by site, which may suggest that sites have differing functional group organization. The results also showed that there was a temporal shift in functional group organization. This can be explained by emergence of insects and temporal niches which vary throughout the growing season. This shift was seen in all patch sizes, suggesting that canebrakes may not differ in temporal insect dynamics compared to adjacent bottomland hardwood forests.

Increased isolation of suitable habitat has been shown to have a negative relationship with arthropod diversity (Tschardt et al 2002). MacArthur and Wilson

(1967) state that diversity decreases when islands are further away and more isolated. Our hypothesis was that more isolated cane patches would have lower diversity compared to less isolated patches. Shannon-Weiner Diversity ( $H'$ ) had a significant negative relationship with distance between cane patches, but this relationship was only found in large and intermediate cane patches. Other measures such as evenness and species richness also had a negative relationship with isolation. Abundance had a positive relationship with distance to nearest canebrake which may suggest crowding effect. These trends showed that increased isolation of canebrakes may have a negative affect on diversity. However, arthropod communities in isolated patches were similar in terms of community similarity compared to less isolated patches.

Fluctuations in diversity and trophic levels have been shown to increase in fragments compared to continuous landscapes (Laurence 2002; Terborgh et al 2001). Laurence (2002) states that hyperdynamism of smaller fragments undergo one of two processes: one having invariably higher dynamics than intact habitats and the other process which begins with higher dynamics in fragments compared to intact habitat followed by gradual stabilization to near pre-fragmented condition. It was hypothesized that smaller patches would have higher fluctuations in diversity and functional group organization than larger patches. Normal fluctuations in arthropod diversity and functional group organization occur because of Arthropod emergence and temporal niches throughout the growing season. The data suggested that smaller cane patches did not have higher fluctuations in diversity and functional group organization compared to larger cane patches and non cane patches. The present results suggest that these



canebrake fragments may have stabilized to near pre-fragmented condition, or a new equilibrium has been reached .

The present research shows that fragmented canebrakes function in terms of diversity, functional group organization, and community composition, much like the adjacent bottomland hardwood matrix. The surrounding matrix does not show any evidence that it is inhospitable, and is capable of supporting a large proportion of the species within the community.

### **Management Applications**

The present data suggests that small, fragmented canebrakes should be conserved to help sustain the unique Arthropod species that may be associated with canebrakes. Bachman's warbler's extinction may be a result of the loss of these cane-associated insects which may have been a primary food source. Since large continuous canebrakes are uncommon, management of smaller canebrakes could be used to maintain uncommon Arthropods. Restoring small cane patches near existing canebrakes may help maintain these uncommon arthropods from possible localized extinction.

## Literature Cited

- Andren, Henrik. 1994. Effects of Fragmentation on Birds and Mammals in Landscapes with Different Proportions of Suitable Habitat. *Oikos*. 71(3) 355-366
- Baldwin, B.S. et al. 2009. Propagation Methods of Rivercane (*Arundinaria gigantea*). *Castanea*. 74(3) 300-316
- Brantley, C.G., and S.G. Platt. 2001. Canebrake conservation in the Southeastern United States. *Wildlife Society Bulletin*. (29)4:1175-1181
- Butchart, S.M.H., et al. 2010. Global Biodiversity: Indicators of Recent Declines. *Science*. 328:1164-1168
- Campbell, J.N. 1985. The Land of Cane and Clover: Presettlement Vegetation in the So-Called Bluegrass Region of Kentucky. Herbarium Report, University of Kentucky, Lexington, USA.
- Chapin, F.S., et al. 2000. Consequences of Changing Biodiversity. *Nature*. 405:234-242
- Cirtain, M. C., S. B. Franklin, and S. R. Pezeshki. 2004. Effects of nitrogen and moisture regimes on *Arundinaria gigantea* (Walt.) Muhl. seedling growth. *Natural Areas J* 24:251-257.
- Collinge, S.K. 2000. Effects of grassland fragmentation on insect species loss, colonization, and movement patterns. *Ecology*. 81(8): 2221-2226
- Datillo, A.J., and Rhoads, C.C.. 2005. Establishment of the Woody Grass *Arundinaria gigantean* for Riparian Restoration. *Restoration Ecology*. 13(4) 616-622
- Debinski, D.M., and Holt, R.D. 2000. A Survey and Overview of Habitat Fragmentation Experiments. *Landscape Ecology*. 14(2) 342-355
- Didham, R.K. et al. 1996. Insects in Fragmented Forests: A Functional Approach. *Trends in Ecology and Evolution*. 11(6) 255-260
- Didham, R.K. et al. 1998. Beetle Responses to Tropical Forest Fragmentation. *Ecological Monographs*. 68: 295-323
- Ewers, R.M., and Didham, R.K. 2006. Confounding factors in the detection of species response to habitat fragmentation. *Biological Reviews*. 81(1) 117-142

- Gagnon, P.R. and Platt, W.J. 2008. Reproductive and Seedling Ecology of a Semelparous Native Bamboo (*Arundinaria gigantea*, *Poaceae*). *The Journal of the Torrey Botanical Society*. 135(3) 309-316
- Gagnon, P.R. 2009. Fire in the Floodplain Forests in the Southeastern USA: Insights from Disturbance Ecology of Native Bamboo. *Wetlands*. 29(2) 520-526
- Gardinier, E.S., and Olivier, J.M. 2004. Restoration of Bottomland Hardwood Forests in the Lower Mississippi Alluvial Valley. *Restoration of Temperate and Boreal Forests*
- Gibb, H. and Hochuli, D.F. 2002. Habitat Fragmentation in an Urban Environment: Large and Small Fragments Support Different Arthropod Assemblages. *Biological Conservation*. 106 (1) 91-100
- Hamel, P. B. and Hooper R. G.,. 1979. Bachman's Warbler-The Most Critically Endangered. Proc. S. C. Endangered Species Symp. 1976: 164-168
- Harrison, S. and Bruna, E.. 1999. Habitat Fragmentation and Large Scale Conservation: What Do We Know for Sure? *Ecography*. 22(3) 225-232
- Jonhson, K.A., et al. 1996. Biodiversity and the productivity and stability of ecosystems. *Trends in Ecology and Evolution*. 11(9) 373-377
- Jules,E.S., and Shanani, P. 2003. A Broader Ecological Context to Habitat Fragmentation: Why Matrix Habitat is More Important Than We Thought. *Journal of Veg, Sci*. 14: 459-464
- Kareiva, P. 1987. Habitat fragmentation and the stability of predator-prey interactions *Nature*. 26:388-390
- Kastner, J. 2000. The Gullible Genius. *Whole Earth*. 102:18
- Rieske, L.K. and Buss, L. J. 2001. Influence of Site on Diversity on Abundance of Ground and Litter Dwelling Coleoptera in Appalachian Oak-Hickory Forests. *Environmental Entomology*. 225: 484-496
- Krauss, J., Steffan-Dewenter, I., and Tscharatke, T. 2003. Local Species Immigration, Extinction, and Turnover of Butterflies in Relation to Habitat Area and Habitat Isolation. *Oecologia* 137(4) 591-602
- Kupfer, J.A., et al. 2006. Not Seeing the Ocean for the Islands: The Mediating Influence of Matrix-Based Process on Forest Fragmentation Effects. *Global Ecology and Biogeography*. 15:8-20

- Layman, C.A., Quattrochi, J.P., Peyer, C.M., and Allgeier, J.E. 2007. Niche width collapse in a resilient top predator following ecosystem fragmentation. *Ecology Letters*. 10(10) 937-944
- Lovei, G.L, Magura, T., Tothmeresz, B., and Kodobocz, V. 2006. The influence of matrix and edges on species richness patterns of ground beetles (Coleoptera:Carabidae) in habitat islands. *Global Ecology and Biogeography*. 15(3) 283-289
- Martinko, E.A., Hagen, R.H., and Griffith, J.A. 2006. Successional Change in the Insect Community of a Fragmented Landscape. *Landscape Ecology* 21(5) 711-721
- McArthur, R.H. and E.O. Wilson. 1967. *The Theory of Island Biogeography*. Princeton University Press. Princeton, New Jersey, USA
- McCann, K.S. 2000. The Diversity-Stability Debate. *Nature*. 405:228-233
- Miyashita, T., Shinkai, A., and Chida, T. 1998. The Effects of Forest Fragmentation on Web Spider Communities in Urban Areas. *Biological Conservation* 86(3) 357-364
- Naeem, S., et al. 1994. Declining biodiversity can alter the performance of ecosystems. *Nature*. 368: 734
- Noss, R.F., Laroe, E.T., and J.M. Scott. 1995. Endangered ecosystems of the United States: A preliminary assessment of loss and degradation. U.S. Dept. of Interior, Nat. Biol. Serv., Biol. Rept. 28:58 p
- Platt, S.G., and C.G. Brantley. 1997. Canebrakes: An Ecological and Historical Perspective. *Castanea* 62: 1
- Platt, S.G., Brantley, C.G., and Rainwater, T.R. 2001. Canebrake Fauna: Wildlife Diversity in a Critically Endangered Ecosystem. *The Journal of the Elisha Mitchell Scientific Society*. 117(1)1-19
- Polus, E., Vandewoestijne, S., Choutt, J., and Baguette, M. 2007. Tracking The Effects of One Century of Habitat Loss and Fragmentation on Calcareous Grassland Butterfly Communities. *Biodiversity and Conservation* 16(12) 3423-3436
- Preston, F.W. 1962. Canonical Distribution of Commonness and Rarity. *Ecology*. 43: 185-215
- Remsen, J.V. 1986. Was Bachman's Warbler a Bamboo Specialist? *Auk*. 103:216-219
- Roosevelt, T. 1908. In the Louisiana Canebrakes. *Scribners Magazine*. 43:47-60

- Schoonover, J.E. and Williard, K.W.J. 2007. Ground Water Nitrate Reduction in Giant Cane and Forest Riparian Buffer Zones. *Journal of the American Water Resources Association*
- Schorger, A.W., 1955. *The Passenger Pigeon*. University of Wisconsin Press, Madison WI.
- Stewart, M.A. 2007. From King Cane to King Cotton: Razing Cane. *Environmental History*. 12:59-79
- Taki, H., and Keven, P.G. 2007. Does habitat loss affect the communities of plant and insect communities equally in plant-pollinator interactions? Preliminary Findings. *Biodiversity and Conservation*. 16(11) 3147-3161
- Taki, H., and Keven, P.G, Ascher, J.S. 2007. Landscape Effects of Forest Loss in a Pollination System. *Landscape Ecology* 22(10) 1575-1587
- Terbough, J. et al. 2001. Ecological Meltdown in Predator Free Forest Fragments. *Science*. 294 (5548) 1923-1926
- Tilman, D. and Downing, J.A. 1994. Biodiversity and Stability in Grasslands. *Nature*.367:363-365
- Tscharntke, T. et al. 2002. Characteristics of Insect Populations on Habitat Fragments: A Mini Review. *Ecological Research*. 17 (2) 229-239
- Valladares, G., Salvo, A., and Cagnolo, L. 2006. Habitat Fragmentation Effects on Trophic Processes of Insect-Plant Food Webs. *Conservation Biology*. 20(1) 212-217
- Ward, D.F, New, T.R., and Yen, A.C. 2001. Effects of pitfall trap spacing on the abundance, richness, and composition of invertebrates catches. *Journal of Insect Conservation*. 5:47-53
- West, E.M. 1934. Canebrakes of the Southeastern U.S. Ph D Distertation. Ohio State University, Columbus
- Worm, B., and Duffy, J.E. 2003. Biodiversity, Productivity, and Stability in Real Food Webs. *Trends in Ecology and Evolution*. 18(12) 628-633
- Yachi, S., and Loreau, M. 1999. Biodiversity and Ecosystem Productivity in a Fluctuating Environment: The Insurance Hypothesis. *Proceedings in the National Academy of Science*. 96(4) 1463-1468

Yahner, R.H. 1988. Changes in Wildlife Communities Near Edges. *Conservation Biology*. 2(4) 333-339

**Table 1.** Canebrake patch area and distance to nearest patch at each study site.

Site	Large		Intermediate		Small	
	Area (ha.)	Distance (m)	Area (ha.)	Distance (m)	Area (ha.)	Distance (m)
River Road	1.15	60	0.42	40	0.04	60
Collierville	0.56	60	0.24	140	0.03	60
Cache River	1.39	40	0.2	30	0.06	30
Dahomey	1.08	30	0.36	80	0.06	15

**Table 2.** Collection dates for each site.

Site	Collection		
	1	2	3
Cache River	7/15/2008	8/6/2008	10/13/2008
Dahomey	6/20/2008	6/26/2008	8/14/2008
Collierville	5/29/2008	7/19/2008	10/20/2008
River Road	6/4/2008	7/6/2008	8/28/2008

**Table 3.** GLM (Type III Sums of Squares) and GLM MANOVA results of Arthropod community diversity variables. Statistical significant F-values indicated by asterisks: \* =  $P < 0.05$ , \*\*\* =  $P < 0.01$ . Abbreviations: Patch = Patch size; Coll. Period = Collection Period.

Effect	GLM Results				GLM MANOVA Results	
	Num d.f.	Richness F-value	Abundance F-value	Diversity F-value	Evenness F-value	Pillai's Trace F-value
Patch	3	1.365	1.24	0.83	0.742	0.866
Coll. Period	2	2.216	8.694***	0.169	0.748	5.752***
Site	3	1.375	7.432*	2.420	2.661	2.010
Patch*Site	9	3.289	0.652	0.602	0.292	1.084
Coll. Period*Site	6	5.357*	4.803*	0.897	1.571	2.93***
Patch*Coll. Period	6	1.958	0.811	0.560	0.376	1.084

**Table 4.** Results of MRPP analysis of patch size and collection and Indicator Species Analysis of functional groups in native canebrakes and adjacent bottomland hardwood forest. *Delta* is the frequency of Type I error and *P* is the proportion of smaller deltas in the MRPP section of the table. Higher IV values indicate that one group is more common at more study sites.

MRPP				
	Patch Size		Collection Period	
	Delta	P	Delta	P
	0.3828	0.851	0.3279	< 0.0001
Indicator Species Analysis				
	I.V.	P	I.V.	P
Phytophagus	54.1	0.09	48.3	0.57
Predator	29.0	0.97	44.5	0.31
Parasite	29.4	0.74	42.7	0.18
Scavenger	33.5	0.64	<b>58.3</b>	<b>0.0004</b>
Omnivore	36.5	0.45	47.3	0.23



**Table 5.** Indicator Species Analysis for the 20 most abundant species found at each site. Higher IV values indicate that one group is more common at a particular study site. Significant P values are in bold.

Species	Percent of Relative Abundance at each patch					IV	Group Max	p
	Large	Intermediate	Small	Non				
Actinedida	40	30	15	15	9.5	2	0.599	
Chordeumida	11	0	26	63	5.6	4	<b>0.001</b>	
Entomobryidae	24	28	30	19	7.9	2	0.249	
Hypogastruridae	22	32	25	21	6.9	2	0.458	
Isotomidae	35	32	18	14	19.6	1	<b>0.027</b>	
<i>Lasius neoniger</i>	22	16	42	21	7.8	3	<b>0.008</b>	
<i>Lasius umbratus</i>	44	8	34	14	7.5	1	<b>0.022</b>	
<i>Leptocera spp.</i>	22	31	17	31	7.3	2	0.121	
Listrophoridae	13	52	16	19	7.5	2	<b>0.001</b>	
<i>Lycosa spp.</i>	27	24	18	30	5.5	4	0.301	
Mymaridae	6	37	29	27	4.7	2	<b>0.029</b>	
<i>Myrmecina americana</i>	34	22	25	19	7.4	1	0.044	
<i>Myrmica punctiventris</i>	30	35	19	16	7.6	2	<b>0.018</b>	
<i>Nemobius carolinensis</i>	32	17	18	30	9.1	1	0.106	
Onychiuridae	0	0	100	0	4.7	3	<b>0.001</b>	
Oribatidae	19	25	25	32	7.4	4	0.577	
Sminthuridae	40	17	23	20	17.6	1	<b>0.001</b>	
<i>Stelidota geminata</i>	25	13	25	37	5.7	4	0.097	
<i>Temnothorax ambiguus</i>	17	21	31	31	5.3	3	0.543	
<i>Tetramorium caespitum</i>	14	27	15	45	9.7	4	<b>0.002</b>	
Torymidae	17	36	24	23	4.9	2	0.091	

**Table 6. Community Similarity of all patch sizes at each collection period at each site**

		Patch Size										
		Large			Intermediate			Small			Non Cane	
		1	2	3	1	2	3	1	2	3	1	2
Large	1	60.27	32.37	57.95	86.13	32.1	65.43	53.12	23.13	57.78	54.04	52.42
	2	60.27	39.74	48.46	40.31	100	46.27	43.8	26.94	47.58	42.24	42.05
	3	32.37	39.74	34.87	20.79	61.55	33.78	45.77	72.29	35.38	32.78	67.77
Intermediate	1	57.95	48.46	34.87	29.93	41.93	67.07	28.22	30.58	69.94	67.13	56.65
	2	86.13	40.31	20.79	29.93	24.03	67.65	28.22	17.87	57.37	49.74	54.19
	3	32.1	100	61.55	24.03	35.44	17.86	65.73	35.38	32.78	67.77	
Small	1	65.43	46.27	67.07	33.66	35.44	63.08	27.82	68.49	65.4	61.35	
	2	53.12	43.8	67.65	28.22	17.86	63.08	37.43	68.49	62.55	59.84	
	3	23.13	26.94	30.58	17.86	65.73	27.82	37.43	65.4	24.74	60.9	
Non Cane	1	57.78	47.58	69.94	57.37	35.38	68.49	68.49	65.4	67.39	52.41	
	2	54.04	42.24	67.13	49.74	32.78	65.4	62.55	24.74	67.39	47.7	
	3	52.42	42.05	56.65	54.19	67.77	61.35	59.84	60.9	52.41	47.7	

# Cache River

	Patch Size												Non Cane	
	Large			Intermediate			Small			Non Cane				
	1	2	3	1	2	3	1	2	3	1	2	3		
Large														
1	54.43	30.67	56.94	55.9	23.28	54.73	60	29.16	62.32	53.79	22.96			
2	54.43	21.27	34.8	69.92	16.21	37.67	77.85	22.92	45.96	78.44	15.99			
3	30.67	21.27	40.74	34.1	68.9	36.81	22.78	70.52	28.06	20.43	72.23			
Intermediate														
1	56.94	34.8	40.74	48.74	39.09	63.08	38.76	44.63	48.52	32.5	36.93			
2	55.9	69.92	34.1	48.74	32.97	43.39	71.5	35.33	51.45	64.47	28.66			
3	23.28	16.21	68.9	32.97	26.34		17.72	64.44	21.31	14.89	71.45			
Small														
1	54.73	37.67	63.08	43.39	26.34		40.73	32.97	58.94	32.85	27.62			
2	60	77.85	22.78	71.5	17.72	40.73	23.89	23.89	52.71	73.33	16.71			
3	29.16	22.92	70.52	35.33	64.44	32.97	23.89		30.39	21.97	62.52			
Non Cane														
1	62.32	45.96	28.06	51.45	21.31	58.94	52.71	30.39		43.33	23.37			
2	53.79	78.44	20.43	64.47	14.89	32.85	73.33	21.97	43.33		15.77			
3	22.96	15.99	72.23	28.66	71.45	27.62	16.71	62.52	23.37	15.77				

# River Road

	Patch Size												Non Cane		
	Large			Intermediate			Small			Non Cane					
	1	2	3	1	2	3	1	2	3	1	2	3			
Large															
1	40.67	56.16	35.4	27.36	27.7	38.08	34.77	35.03	60.83	33.56	29.3				
2	40.67	36.83	32.22	44.85	34.42	54.99	42.31	28.43	38	45.99	32.85				
3	56.16	36.83	35.02	40.42	53.46	33.11	34.73	55.73	53.55	44.82	53.35				
Intermediate															
1	35.4	32.22	35.02	43.51	41.82	43.72	40.56	29.8	55.54	55.14	39.5				
2	27.36	44.85	40.42	43.51	41.4	36.6	44.94	30.15	38.54	54.37	39.41				
3	27.7	34.42	53.46	41.82	41.4	35.16	35.17	42.78	32.13	43.91	65.18				
Small															
1	38.08	54.99	33.11	36.6	35.16	35.58	35.58	18.6	49.45	54.43	27.89				
2	34.77	42.31	34.73	44.94	35.17	35.58	29.9	40.11	47.55	33.98					
3	35.03	28.43	55.73	30.15	44.94	18.6	29.9	43.9	31.65	46.27					
Non Cane															
1	60.83	38	53.55	38.54	32.13	49.45	40.11	43.9	52.83	33.4					
2	33.56	45.99	44.82	54.37	43.91	54.43	47.55	31.65	52.83	39.86					
3	29.3	32.85	53.35	39.41	65.18	27.89	33.98	46.27	33.4	39.86					

# Collierville

	Patch Size												Non Cane		
	Large			Intermediate			Small			Non Cane					
	1	2	3	1	2	3	1	2	3	1	2	3			
Large															
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
	40.01	40.01	40.4	63.12	42.14	36.04	60.07	48.05	45.64	65.35	57.29	25.88	40.01	40.01	40.4
	40.4	25.91	25.91	43.53	78.84	43.33	33.35	59.01	25.32	41.4	44.85	19.82	40.4	25.91	25.91
	63.12	43.53	28.17	28.17	26.49	55.8	39.8	28.46	69.92	32.18	29.58	44.76	63.12	43.53	28.17
Intermediate															
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
	42.14	78.84	46.69	46.69	43.37	26.21	34.21	59.77	25.76	44.13	55.73	19.42	42.14	78.84	46.69
	36.04	43.33	55.8	45.74	43.37	26.21	26.21	45.71	58.16	34.96	28.18	19.42	36.04	43.33	55.8
	60.07	33.35	39.8	55	34.21	26.21	26.21	38.62	34.32	59.2	54.09	18.1	60.07	33.35	39.8
	48.05	59.01	28.46	49.98	59.77	45.71	38.62	59.77	28.7	49.68	52.01	23.99	48.05	59.01	28.46
	45.64	25.32	69.92	34.93	25.76	58.16	34.32	28.7	29.82	24.8	60.97	23.2	45.64	25.32	69.92
	65.35	41.4	32.18	58.89	44.13	34.96	59.2	49.68	24.8	60.97	17.7		65.35	41.4	32.18
	57.29	44.85	29.58	48.92	55.73	28.18	54.09	52.01	24.8	60.97			57.29	44.85	29.58
	25.88	19.82	44.76	27.52	19.42	54.57	18.1	23.99	61.12	23.2	17.7		25.88	19.82	44.76
Small															
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
	40.01	40.01	40.4	63.12	42.14	36.04	60.07	48.05	45.64	65.35	57.29	25.88	40.01	40.01	40.4
	40.4	25.91	25.91	43.53	78.84	43.33	33.35	59.01	25.32	41.4	44.85	19.82	40.4	25.91	25.91
	63.12	43.53	28.17	28.17	26.49	55.8	39.8	28.46	69.92	32.18	29.58	44.76	63.12	43.53	28.17
Non Cane															
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
	40.01	40.01	40.4	63.12	42.14	36.04	60.07	48.05	45.64	65.35	57.29	25.88	40.01	40.01	40.4
	40.4	25.91	25.91	43.53	78.84	43.33	33.35	59.01	25.32	41.4	44.85	19.82	40.4	25.91	25.91
	63.12	43.53	28.17	28.17	26.49	55.8	39.8	28.46	69.92	32.18	29.58	44.76	63.12	43.53	28.17

**Table 7.** Repeated Measures ANOVA and MANOVA results of Arthropod community response variables. Statistical significant F-values indicated by bold lettering and asterisks: \* =  $P < 0.05$ , \*\* =  $P < 0.01$ . Abbreviations: Patch = Patch size; Coll. Period = Collection Period.

MANOVA Results					
Pillai's Trace					
Effect	Num d.f.	Richness	Abundance	Diversity	Evenness
Coll. Period	2	1.21	<b>10.64**</b>	1.98	3.36
Coll. Period *Site	6	1.27	<b>3.39*</b>	2.45	<b>2.90*</b>
Coll. Period *Patch	6	0.65	0.57	0.29	0.39

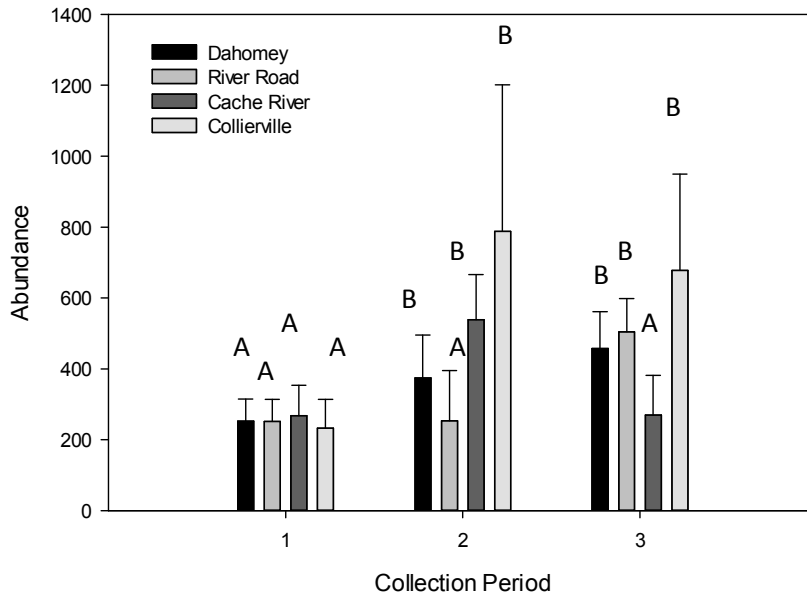
Repeated Measures ANOVA results					
	Num d.f.	Richness	Abundance	Diversity	Evenness
		F-value	F-value	F-value	F-value
Patch	3	0.35	1.92	1.3	1.83
Coll. Period	2	1.70	<b>7.99**</b>	2.07	<b>4.05*</b>
Site	3	1.06	<b>8.57**</b>	3.240	<b>5.12*</b>
Coll. Period * Site	6	1.82	<b>3.46*</b>	2.45	<b>4.32**</b>
Patch * Coll. Period	6	0.78	0.82	0.29	0.48

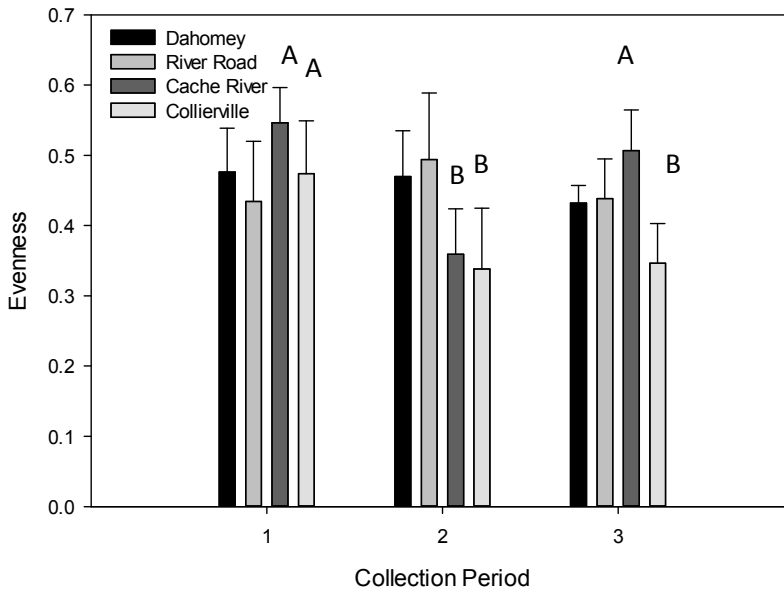
Repeated Measures ANOVA results at each collection period					
	Num d.f.	Richness	Abundance	Diversity	Evenness
		F-value	F-value	F-value	F-value
<i>Coll. Period 1</i>					
Patch Size	3	0.29	0.68	2.35	0.59
Site	3	1.71	0.13	0.38	1.35
<i>Coll. Period 2</i>					
Patch Size	3	0.93	1.74	3.21	1.2
Site	3	0.63	<b>6.238*</b>	0.7	<b>6.73*</b>
<i>Coll. Period 3</i>					
Patch Size	3	0.46	0.27	1.53	1.08
Site	3	2.21	3.32	<b>6.51*</b>	<b>6.68*</b>

**Figure 1.** a) Total abundance of Arthropods at each site and collection period. b) Total evenness at each site and collection period. Letters indicate differences detected by Tukey's post hoc test

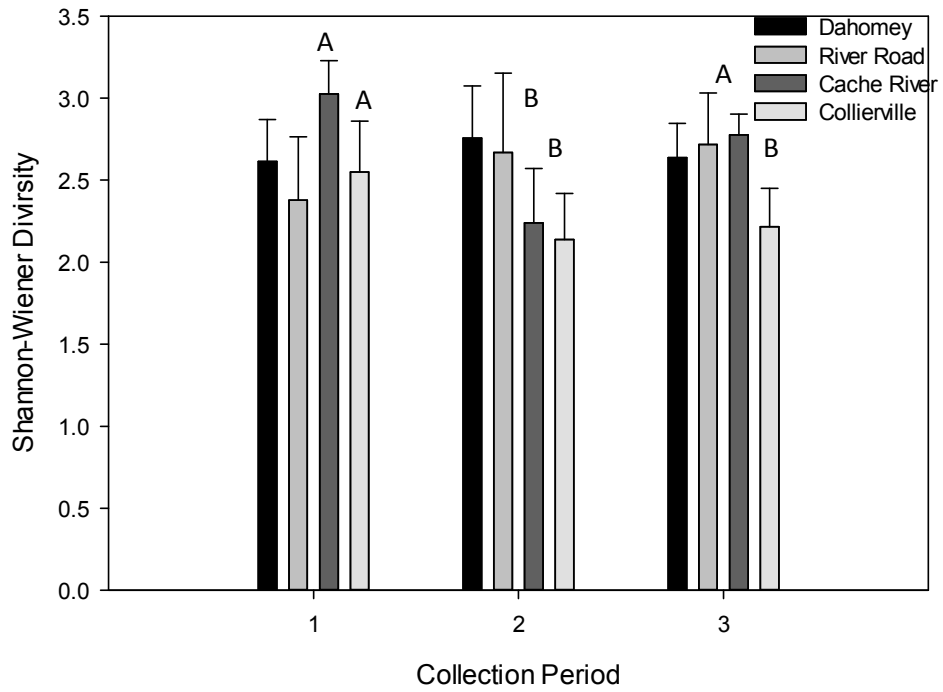
**a)**



**b)**



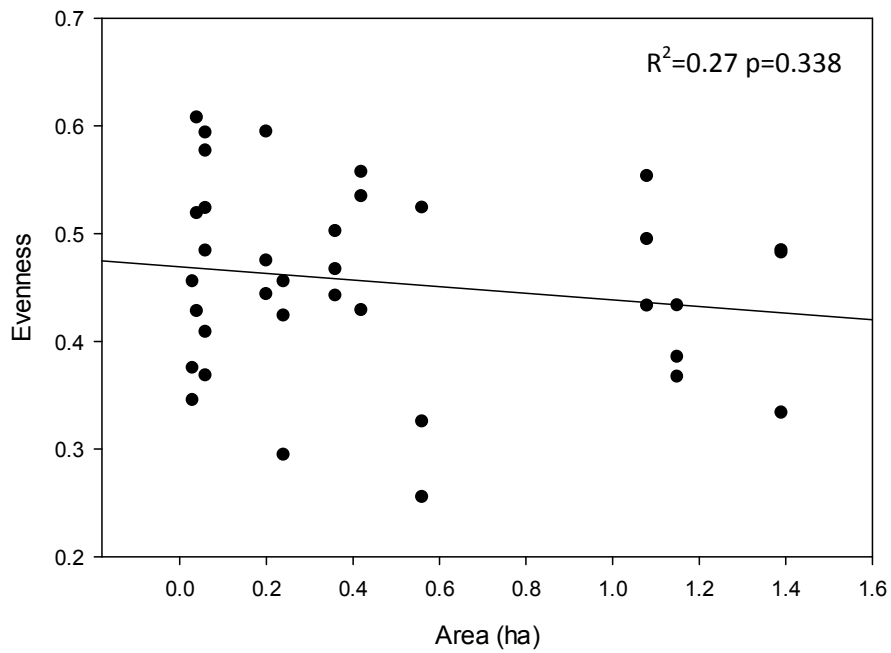
**Figure 2.** Total Shannon-Wiener Diversity ( $H'$ ) at each site and collection period. Letters indicate pair-wise differences detected by Tukey's post hoc.



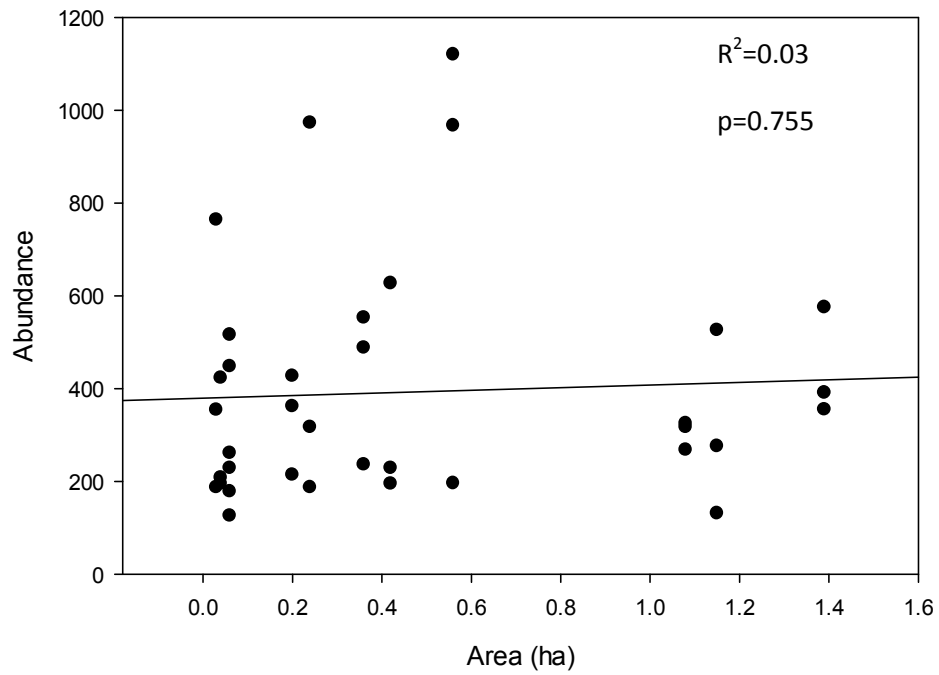


**Figure 3. a)** Linear Regression of the relationship between **a)** evenness and **b)** abundance at various canebrake patches.

**a)**

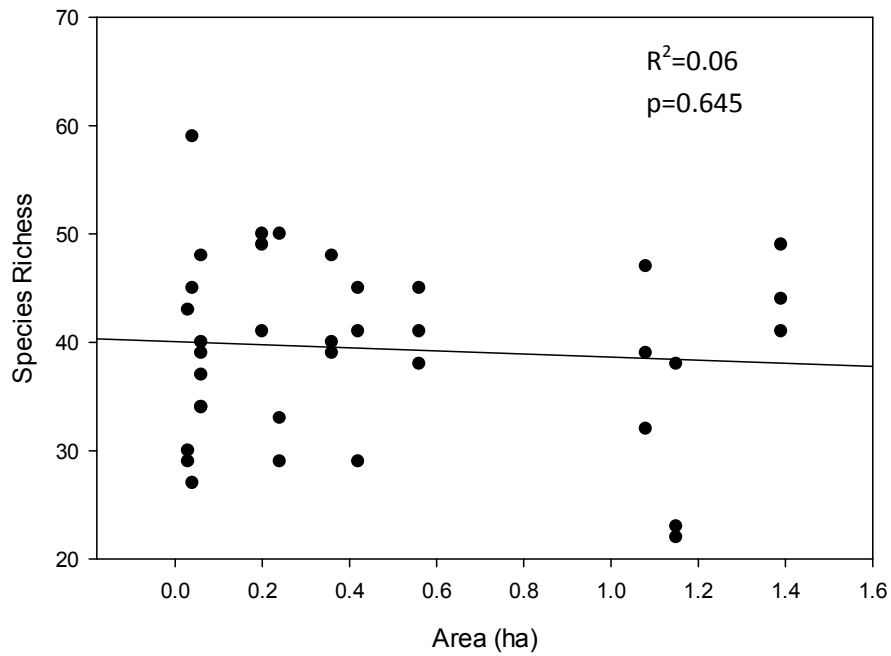


**b)**

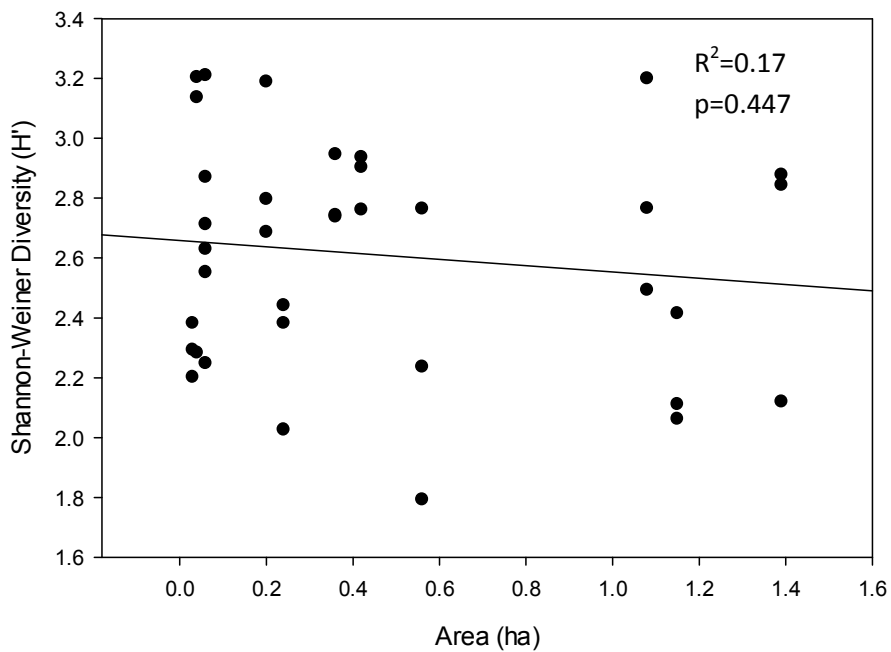


**Figure 4.** Linear Regression of the relationship between **a)** species richness and **b)** Shannon-Wiener Diversity at various canebrake patches

**a)**

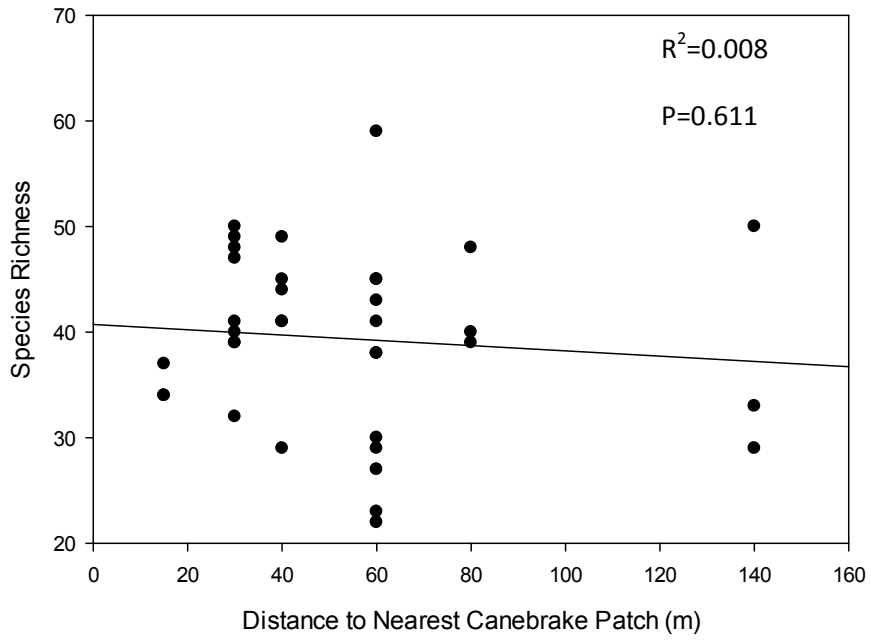


b)

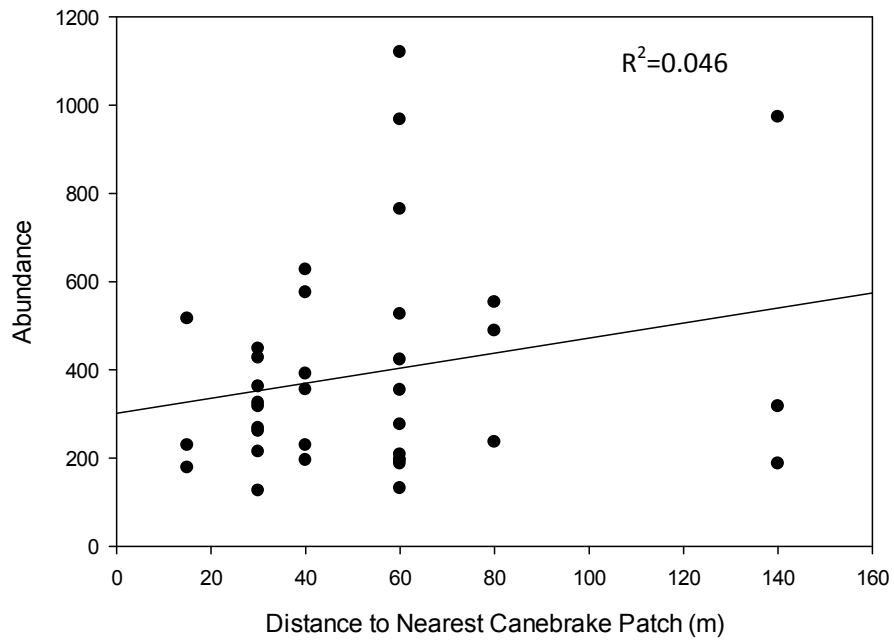


**Figure 5.** Linear Regression of the relationship between a) species richness and b) abundance with the distance to nearest canebrake in all cane patches.

a)

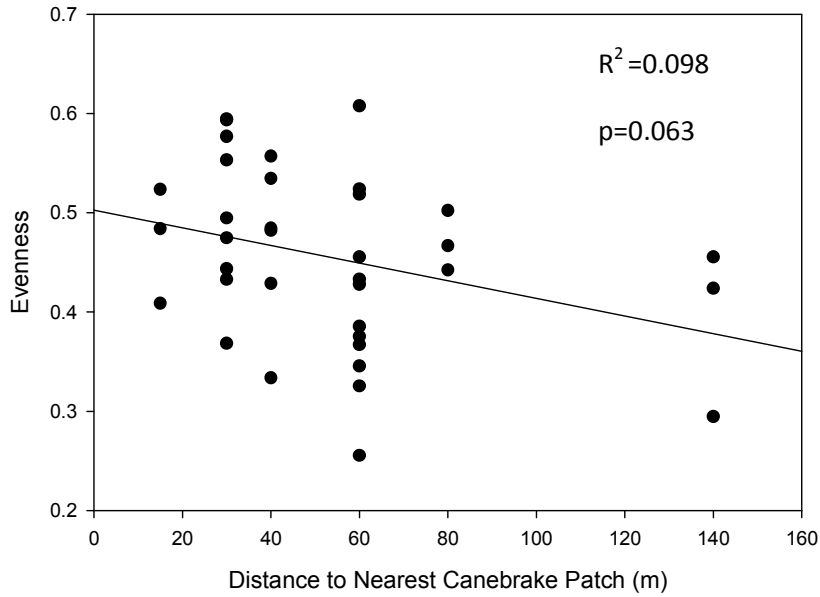


b)

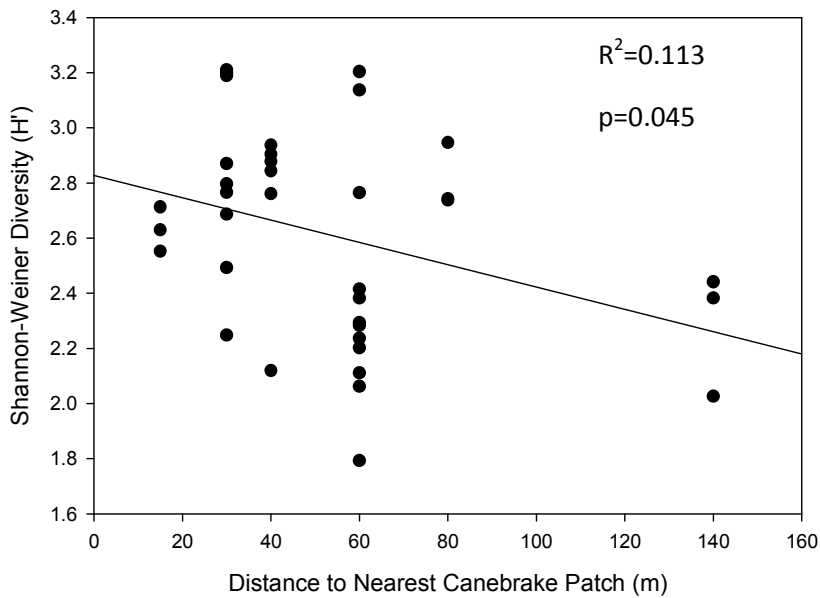


**Figure 6.** Linear Regression of the relationship between **a)** evenness and **b)** Shannon-Wiener Diversity and distance to nearest canebrake in all cane patches. ( $R^2=0.113$ ,  $F=4.349$ ,  $p=0.045$ )

**a)**

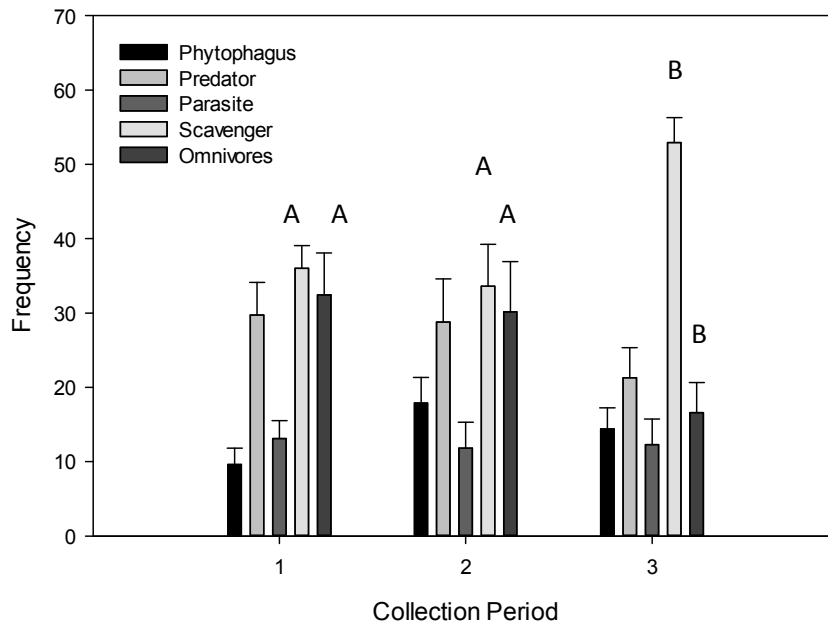


**b)**

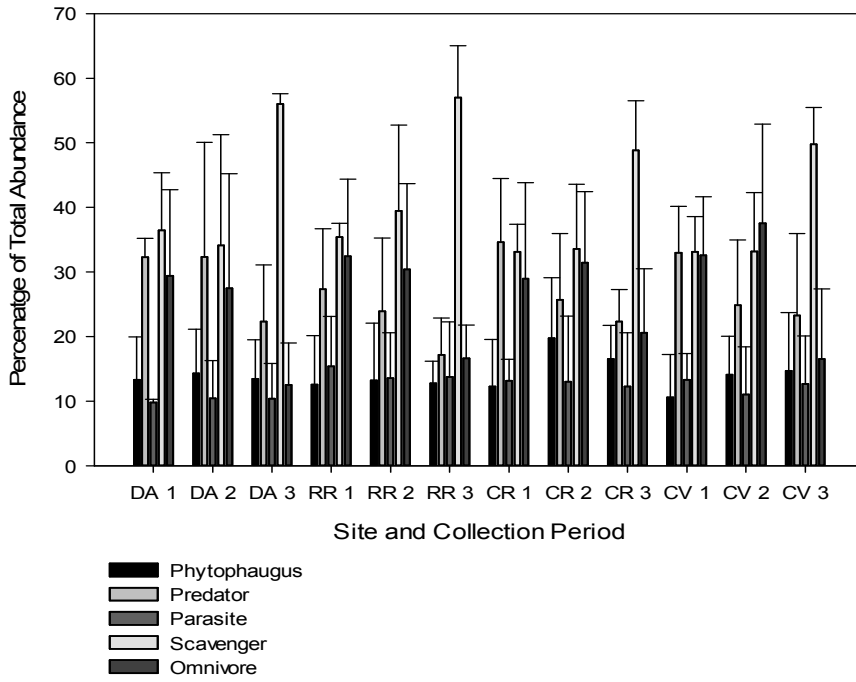


**Figure 7. a)** A comparison of functional group organization at each collection period. **b)** A comparison of functional group organization all patch sizes at all site and all collection periods. Letters indicate significant Tukey's pair wise differences.

**a)**

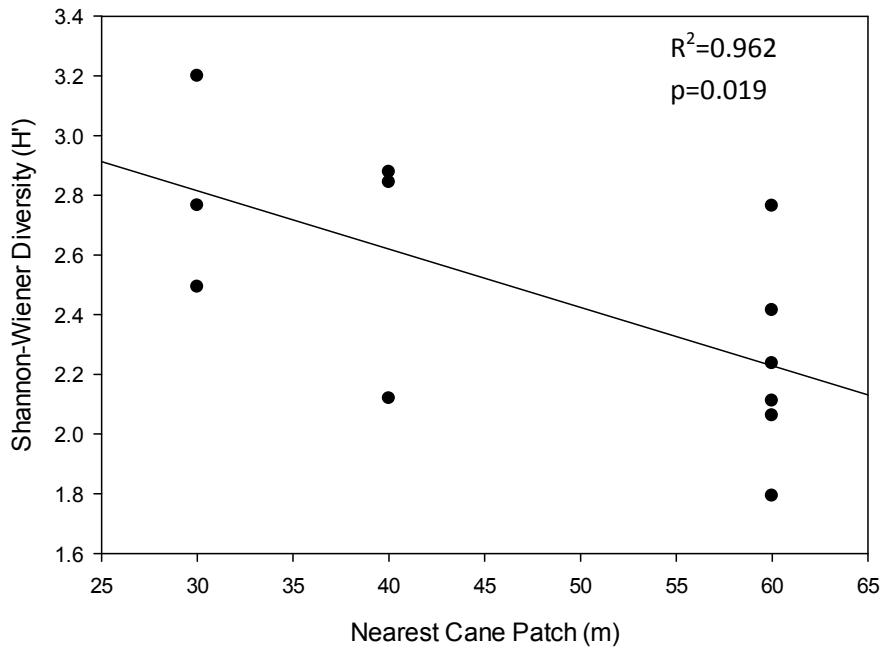


**b)**

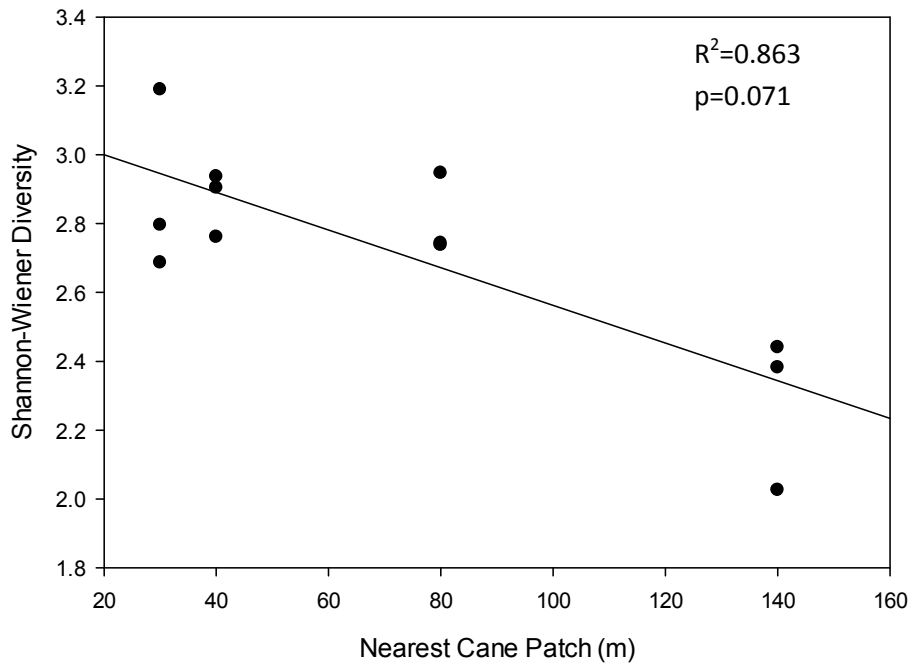


**Figure 8. a)** Linear regression of the relationship between Shannon-Wiener diversity with distance to nearest patch in **a)** large cane patches ( $R^2=0.962$ ,  $F=51.208$ ,  $p=0.019$ ) and **b)** intermediate cane patches ( $R^2=0.863$ ,  $F=12.563$ ,  $p=0.071$ )

**a)**



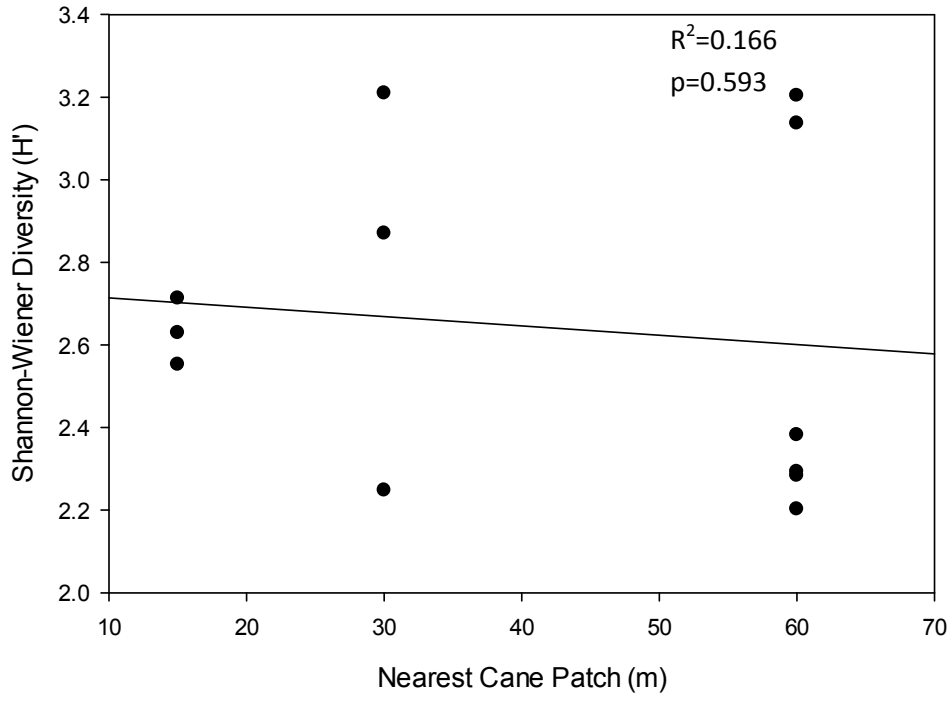
b)



**Figure 9. a)** Linear Regression of the relationship between Shannon-Wiener Diversity with distance to nearest patch in small cane patches( $R^2=0.166$ ,  $F=0.398$ ,  $p=0.593$ )

a)





**Table 8.** Species occurrence and abundance at canebrake patches and adjacent bottomland hardwood forest at all four study sites.  
L=Large Cane; I=Intermediate Cane; S=Small Cane; N=Non Cane

Species	Dahomey				River Road				Cache River				Collierville				
	L	I	S	N	L	I	S	N	L	I	S	N	L	I	S	N	
<i>Abacidus permundus</i>	0	2	1	4	0	6	1	0	0	0	0	0	0	0	0	0	0
<i>Acanaloniidae</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Actineda</i>	10	80	53	40	0	0	0	0	31	42	6	20	182	42	25	20	20
<i>Admonita sp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Aenigmatias sp.</i>	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0
<i>Agritotes oblongicollis</i>	2	1	1	3	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Alaus oculatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Alebra sp.</i>	0	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0
<i>Alleculidae</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Allocosa sp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>Amara exarata</i>	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
<i>Anomala innuba</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0
<i>Anomoglossus emarginatus</i>	0	0	0	0	2	0	0	0	0	0	1	0	0	0	0	0	0
<i>Anopheles quadrimaculatus</i>	0	0	1	1	0	0	0	2	0	5	4	7	0	1	0	0	0
<i>Anthocoridae</i>	0	21	59	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Anurogryllus muticatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1















<i>Gonatopus</i> spp.	11	5	0	1	0	0	2	0	2	2	1	0	0	4	0	0	1
<i>Graciidae</i>	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Gracilaridae</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Graphocephala coccinea</i>	0	0	0	0	0	0	0	0	3	1	0	0	0	0	0	0	0
<i>Gryllidae</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Gryllus domesticus</i>	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0
<i>Gryllus neglectus</i>	0	0	0	0	0	0	2	0	2	1	0	0	0	0	0	0	0
<i>Habrocestum pulex</i>	0	0	0	0	0	0	0	0	2	0	0	1	0	0	0	0	0
<i>Habrocestum</i> sp.	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Habronattus</i> spp.	0	0	0	1	0	0	1	0	0	0	0	0	0	0	1	1	0
<i>Hahnia</i> sp.	3	3	2	1	0	0	0	1	1	1	1	1	0	0	0	0	0
<i>Hapithus agitator</i>	0	0	0	0	0	0	0	0	1	1	1	4	1	1	3	0	0
<i>Haplodrassus</i> sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Harpalus compar</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
<i>Harpalus erythropus</i>	0	0	0	0	0	2	1	1	0	0	0	0	0	0	0	0	0
<i>Heilius</i> sp.	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Hemerobiidae</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Herpyllus</i> sp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
<i>Hister abbreviatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Hister interuptus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Holothrips</i> sp.	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0









<i>Orchestina</i> sp.	10	20	17	23	13	7	28	28	25	48	15	10	54	50	73	112
<i>Ormyridae</i>	1	0	0	0	0	0	0	0	2	3	5	3	0	1	2	2
<i>Orocharis saltator</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0
<i>Omosita colon</i>	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	2
<i>Orsodacne atra</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Oxycera</i> sp.	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0
<i>Pangaeus</i> spp.	6	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Parandrinae</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
<i>Parcoblatta pennsylvanica</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Pardosa</i> sp.	20	5	1	4	1	1	1	3	1	2	5	15	0	2	0	0
<i>Patrobis longicornis</i>	0	0	0	0	0	2	0	0	0	2	1	2	0	1	0	0
<i>Pentaria trifasciatus</i>	0	0	0	0	0	0	0	0	5	2	1	2	0	0	0	1
<i>Penthaleidae</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Phalacrocera</i> sp	18	12	9	3	0	0	0	0	85	11	62	77	0	0	28	15
<i>Phalangida</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
<i>Philaenus</i> sp.	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Philonthus blandus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
<i>Philonthus fusiformis</i>	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
<i>Philonthus lomatus</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>Philonthus politus</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0



<i>Pterostichus adoxus</i>	0	0	0	1	1	2	2	9	5	10	5	13	4	9	17
<i>Pyraidae</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	0	13
<i>Pyramica membranifera</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
<i>Rembus laticollis</i>	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
<i>Rileyinae</i>	5	1	2	2	4	4	4	6	3	4	10	12	7	11	11
<i>Romalea microptera</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Rophitinae</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Rugilus angularis</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>Rugilus sp.</i>	0	0	0	0	0	0	0	0	2	0	0	1	0	0	0
<i>Saprinus assimilis</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Sassacus sp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Scaphisoma sp.</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>Scarabiidae</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Scarites substriatus</i>	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0
<i>Scarites subterraneus</i>	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0
<i>Scelionidae</i>	0	3	1	3	0	0	0	0	0	0	0	0	0	0	0
<i>Schizocosa sp.</i>	0	0	0	3	0	2	4	1	1	0	0	1	2	1	1
<i>Schizopteridae</i>	0	0	0	0	0	5	0	0	0	0	1	1	0	0	1
<i>Sciara sp.</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>Scudderia septentrionalis</i>	0	0	0	0	0	0	0	7	2	5	1	27	7	62	27
<i>Senotainia spp.</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0









