

EFFECTS OF FIRE FREQUENCY AND THE RED IMPORTED FIRE ANT
ON INSECTS IN A LOUISIANA LONGLEAF PINE SAVANNA

A Dissertation

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ABSTRACT

Ants and ground beetles were collected in pitfall traps from a longleaf pine savanna in the early stages of restoration. Insect abundance and species richness were compared among four treatment combinations of two fire frequencies and presence and absence of an exotic ant, *Solenopsis invicta*, the red imported fire ant. Amdro[®] was used to suppress *S. invicta*. Fire treatments consisted of a single fire in a two year period or two fires in a two year period, one applied each year. Fires were applied late in the growing season both years.

Ten of the 28 ground beetle species collected were single individuals. Samples were not large enough analyze statistically. Burning a fire-maintained habitat does not typically alter ground beetle populations, but there were not enough data to validate this claim.

Twenty-six species of ants were collected from pitfall traps. Six species were exotic. Exotics comprised 23% of the species richness and 98% of the individuals collected. *Solenopsis invicta* far exceeded other species' abundances and accounted for 95% of total ants collected. Excluding *S. invicta*, natives and exotics were relatively equal in abundance. Regression analyses of species ranks and abundances were not different among treatments. Species composition among treatments was more similar following treatment applications.

After many years of fire suppression, many ant species increased in abundance following the first applied fire. However, the second applied fire did not elicit the same magnitude of response. Soil moisture, soil temperature, and leaf litter dry weights were not correlated with ant abundance. Because there were no differences in native

abundance among treatments, it was concluded that Amdro[®] was not necessary to maintain a balance between *S. invicta* and native ants. A single applied fire without Amdro[®] produced the same results with respect to *S. invicta* and native ant abundance as two fires with Amdro[®]. The greatest number of exotic ants were collected from treatments with two fires and no Amdro[®], suggesting more frequent fires increase exotic ant abundance. Data suggest that native ants are able to coexist with *S. invicta* and other exotics even when vastly outnumbered by them.

INTRODUCTION

Longleaf pine savannas possess some of the greatest diversity of herbaceous plants of any ecosystem. More than 200 species of plants exist within individual longleaf pine savannas, including over 130 endangered or threatened species (Appendix 1) (Noss 1995, Noss et al. 1995, Outcalt and Sheffield 1996)). The peril of so many species has come about from large reductions in the vast area the longleaf pine ecosystem once covered and a “geological history that fostered evolution of a narrow endemic taxa, and staggering loss of these communities from agriculture, plantation forestry, and fire suppression” (Noss et al. 1995). This habitat once covered more than 24 million hectares in the southeast, today, there is less than 2% remaining in the form of fragments and degraded remnants (Noss et al. 1995). In Louisiana there has been a 95-99% loss of longleaf pine habitat in the eastern portion of the state and a 75-90% loss in western Louisiana (Noss et al. 1995). Seventy-five percent of all longleaf pine stands are 40.5 hectares or less in size (Outcalt and Sheffield 1996). Louisiana has approximately 94,292 hectares with most trees less than 100 years of age (Outcalt and Sheffield 1996). Longleaf pine forests and savannas are classified as a critically endangered ecosystem (Noss et al. 1995).

Longleaf Pine Ecosystem - Description

Most flatwoods pine savannas occur on the Gulf Coastal Plain, although the longleaf pine ecosystem as a whole extended from Virginia to Texas (Platt 1999). The soil in the Gulf Coastal Plain consists of sand, loam, and clay. When clay is near to the surface soil remains saturated, creating bogs. These soils are nutrient poor (low in N and P), which creates open grassy areas called savannas with scattered clusters of pines. One

difference between flatwoods savannas and grasslands is that the former contain woody vegetation, usually sporadically distributed (Oosterheld et al. 1999). Even though savannas look fairly flat, there is sufficient slope to create differences in hydrology that influence community structure (Oosterheld et al. 1999). Bogs are located in low areas and pines along rises. The combination of soil type, bogs, and low nutrient levels makes longleaf pine savannas suitable for many carnivorous plants such as pitcher plants, venus flytraps, sundews, bladderworts, and butterworts (Platt 1999).

Fires retard growth of woody shrubs and preserve diversity of native groundcover species. Longleaf pine savannas exist as a "fire climax community" (sensu Champman 1932) so long as fires occur frequently. Longleaf pine (*Pinus palustris*) and many other inhabitants of a savanna community are fire adapted (Platt 1999). Timing and frequency of fires are important for eliminating unwanted species and promoting native species (Lovell et al. 1982). Kalisz and Stone (1984) found that historically, longleaf pine habitat in Florida burned naturally every two to five years. While researchers and managers realize the importance of mimicking natural events, the trend for managing longleaf pine savannas is "not to recreate historic fire regimes, but rather ecological results of those regimes" (Simmons et al. 1995).

Longleaf Pine Ecosystem - History

For over 20,000 years pines and oaks have alternated dominance in the southeastern portion of the U.S. (Platt 1999). Though the historic range for *P. palustris* encompassed 37 million hectares, today 75% of all longleaf pine stands are 40.5 hectares or less in size (Outcalt and Sheffield 1996). The start of this devastation can be traced back to the time period (1607-1930) when naval stores such as turpentine, pitch, and

rosin were extracted from longleaf pine. One barrel was the equivalent of 33 trees (Frost 1993). In the 1800's longleaf pines were logged extensively; first along rivers and then with the advent of the railroad virtually every longleaf pine habitat was accessible (Outcalt and Sheffield 1996). Next came livestock farming; hogs ate seedling roots. During the 1920's an observation that fire destroyed *P. palustris* seedlings fostered a period of fire suppression by the Forest Service (Frost 1993). People soon realized that many longleaf pine stands were not regenerating. Sexually mature, cone producing trees had been logged, seedlings were being grazed, and fire suppression did not allow for any new growth. By 1931 loblolly (*Pinus taeda*) and slash pine (*Pinus elliottii*) plantations replaced much of the former longleaf pine acreage. Finally in 1943, the Forest Service reintroduced fire into forest management (Frost 1993).

Longleaf Pine Ecosystem – Recovery and Management

Longleaf pine savannas are not considered to be a subclimax stage in succession. They will not become hardwood forests without human intervention (Platt 1994, Platt 1999). Research showed productivity or biomass within a savanna increased linearly with precipitation, and fire frequency also increased with greater annual precipitation (Oosterheld et al. 1999). In a system where disturbance was a naturally occurring phenomenon manipulation of the disturbance as part of the restoration process is important.

Presently, land managers know that periodic fires do not cause major population shifts in native species (Bendell 1974). Inhabitants of a habitat subjected to chronic disturbance are likely to develop adaptations that enhance their survival and future predictability of the disturbance (Platt 1994). For example, *P. palustris* in the seedling or

grass stage of its life cycle are very resistant to damage by periodic fires. Scales and a cluster of needles surround and insulate the terminal bud. Adult pines possess thick, insulating bark and lose their lower branches, features that prevent fires from reaching their crowns (Platt 1999). Ninety-five percent of ground cover plants are perennials with resprouting mechanisms for regeneration after fire (Platt 1999). Grasses and flowers such as wiregrass, bluestem, liatris, and pityopsis produce significantly more flowers after a growing season fire (Platt 1999, Streng et al. 1993). Plant feeding insects increase rapidly after fires due to new plant growth, and birds respond to the increase in insect activity (Chamrad and Dodd 1972, Euler and Thompson 1978, Lussenhop 1976, Willig and McGinley 1999).

Historical records, tree rings, and soil layers indicate savannas burned every three to four years from lightning ignited fires (Chapman 1932, Christensen 1978, Wahlenberg 1946). "Forest fires [were] more common in the region of longleaf pine than in any other portion of the U.S." (Heyward 1939). Most fires occurred during May through August (Platt 1999). Beginning in May and into June the number of rain free days increased, drying out potential fuel (leaf litter, needles). During June and extending through August lightning storms ignited the fuel (Olson and Platt 1995). Needles of longleaf pine are highly flammable when dry, increasing the likelihood of fire. Annual fires are possible every year because there is 100% recovery of preburn biomass on an annual basis (Oosterheld et al. 1999). Researchers have found here in the Southeast, where mean annual precipitation can exceed 700mm, the longleaf pine ecosystem experiences high fire frequency and high annual productivity. In this system, fire can increase productivity

up to 5x the mean (Oosterheld et al. 1999). Fire is fundamental to the health of this ecosystem.

Fire and Insects

Little is known about insects in the longleaf pine ecosystem, even though 70% percent of the flora is pollinated by insects (Folkerts et al. 1993). Four to five thousand insect species maybe associated with this community (Folkerts et al. 1993). Insect populations distributed in remnant habitats are predisposed to extinction if they cannot migrate between fragments (Panzer 1988). Consequently, there are many rare and threatened invertebrates associated with fire maintained ecosystems and unfortunately, not much is known about species-specific responses to fire regimes (Simmons et al. 1995). Abundance studies of insects on burned and unburned plots has shown that insects in habitats with periodic fires are fire adapted, but have varied responses to fire (Anderson et al. 1989).

There seems to be some discrepancy among research findings about the effect of fire on overall abundance of insects. Cancelado and Yonke (1970) found that collections from burned areas were significantly larger than from unburned areas. But Anderson et al. (1989) found that it took two to three years for the abundance of insects on burned sites to match that of unburned sites. More non-fire adapted insects exist on a site the longer the time between fires, so long intervals without fires will produce a more dramatic drop in number of species and individuals after a fire (Anderson et al. 1989). Individual species reactions to a fire are also influenced by indirect effects such as altered vegetation relationships, removal of litter, increased soil temperature, and moisture loss (Anderson et al. 1989, Willig and McGinley 1999). For instance, phytophagous insects

and "weedy" species able to escape fire quickly move back into burned areas because vegetation grows rapidly and competition is at a minimum (Cancelado and Yonke 1970).

Formicidae - Ants

Ants and spiders were two groups caught most often in pitfall traps in Louisiana grasslands and pastures (Howard and Oliver 1978). Ants can have a substantial impact on grassland ecosystems. They consume a disproportionately large amount of food, feeding on items such as seeds, vegetation, arthropods, and other fauna. They can also affect the distribution of other predator and/or ant species by competitive exclusion. Ants are efficient nutrient recyclers. Consequently their mounds are often rich in phosphorous, potassium, and nitrogen compared to the surrounding soil (Anderson 1990, Beattie 1989, Curry 1994). Mounds serve as protection from fire and the environment. However, fire exerts greater indirect effects on ants than it does on other fire adapted prairie insects by creating changes in the habitat that alter food supplies and competitive interactions (Anderson 1991). In a study where ants were surveyed in plots exposed to three different fire regimes, annual, biennial, and unburned for 14 years, differences in ant species and abundances were attributed to structural changes in habitat due to fire, specifically litter accumulation. Sites burned annually had consistently more ants and more species of ants compared to unburned sites. Biennially burned sites were intermediate between the other two regimes (Anderson 1991). An earlier study by Anderson showed that ants were significantly more abundant on burned sites than unburned sites up to one year after the burn (Anderson et al. 1989).

Carabidae – Ground Beetles

Carabid beetles are important predators in grasslands (Morrill 1992), but not all carabids are predators. A number of seed-eating species may potentially inhabit longleaf pine savannas. For example, in the genus *Pterostichus*, 19 species are grass and conifer seed-eaters; *Poecilus* has two conifer seed-eating species; *Harpalus*, 29 grass and conifer seed-eaters, and *Agonum* has 11 grass seed-eaters (Johnson and Cameron 1969).

Phytophagous carabids respond to plant growth, but predaceous carabids respond to abundance of other insects. Leaf litter accumulation increases carabid activity and presence. Litter also increases ground level humidity, which in turn enhances survival of carabid larvae (Lavigne and Campion 1978, Morrill 1992, Rushton et al. 1989).

Solenopsis invicta – Red Imported Fire Ant

Exotic species impact 49% of endangered species through competition and predation (Springett 1976). The red imported fire ant, *Solenopsis invicta*, is native to South America. This species has been suggested to have originated from the Pantanal region (i.e. Paraguay River flood plain) of South America (Buren et al. 1974). It was introduced into the United States through Mobile, Alabama around 1918, but was not recognized as a new exotic until 1930 (Davidson and Stone 1989, Vinson and Sorenson 1986). By the early 1980s *S. invicta* had invaded all southeastern states from Texas to Georgia and north to Arkansas and North Carolina (Camter 1981). By 1999 they had invaded Virginia, Oklahoma and California, infesting over 121 million hectares (Wojcik et al. 2001). They are now costing states in the Southeast approximately \$1 billion/year in damage and control costs (Pimentel et al. 2000). These ants are successful invaders and have many characteristics that allow them to be successful colonizers (Porter and

Savignano 1990). They prefer heavily disturbed habitats, are adaptable to a broad range of environmental conditions, have high reproductive rates, and feed on a wide variety of foods (Orians 1989, Pimentel et al. 2000, Vinson and Greenberg 1986, Whitcomb et al. 1972, Wilson and Oliver 1969). *Solenopsis invicta* is an efficient predator of everything from insects to small mammals, but it also feeds on plants and seeds and is an active scavenger (Whitcomb et al. 1972). *Solenopsis invicta* responds to abundance of a food source rather than diversity (Ali et al. 1984).

Researchers are divided as to the effect that *S. invicta* has on native fauna. *Solenopsis invicta* restructures the ant community by reducing native species' diversity and abundance from competition and predation (Camilo and Philips 1990, Vinson 1994, Whitcomb et al. 1972). Natives like *Solenopsis xyloni* and *Solenopsis geminata*, along with many *Pheidole* species decrease in abundance or are displaced (Camilo and Philips 1990, Wojcik 1994). Similar studies have shown that when *S. invicta* was removed from an area there was a resurgence of other predatory species such as Carabids (Brown and Goyer 1982, Howard and Oliver 1978). Some species coexist or even increase in abundance when *S. invicta* invades, such as *Monomorium minimum*, *Paratrechina arenivaga*, and *Dorymyrmex* species (Camilo and Philips 1990, Wojcik et al. 2001). As of 1990, impacts of the red imported fire ant on plants and animals were still poorly known (Porter and Savignano 1990). Much research has been initiated since then to further our understanding about the impact this species has had. Unfortunately, most biodiversity studies in the southeast are being conducted now that the ants are well established and pre-establishment comparisons are not possible.

Objective

Loss of habitat and introduced species are the leading causes of native species population decline (Bright 1999, Dobson and Czech 1997, Everett 2000, Wilcove et al. 1998). Many endangered species are also threatened by fire suppression in combination with exotic species; there is a need for active management strategies (Wilcove and Chen 1998, Wilcove et al. 1998). Most species in preserved remnants and restored areas are at risk. Costly restoration and scientific uncertainty do not leave room for generalizations when it comes to management. Standards for measuring success of restoration efforts must be site specific (Maina and Howe 2000, Zedler 2000). Insects constitute a large portion of the overall biota in any habitat and are good indicators of restoration effects and ecosystem functions. For ecosystem health and stability, as many components of the system as possible should be included in the maintenance equation. Therefore, the objectives of my study were to document responses of selected native and exotic insect populations (i.e. Formicidae, Carabidae, and Staphylindae) to two fire frequencies, and to record differences in selected native insect populations in the presence and absence of *Solenopsis invicta* in a Louisiana longleaf pine savanna.

METHODS

Study Area

I conducted this study at Lake Ramsay Wildlife Management Area (WMA) and the adjoining Lake Ramsay Preserve owned by The Nature Conservancy of Louisiana (30° 30' N, 90° 10' W) in St. Tammany Parish, Louisiana. Purchased in 1992 and 1994 respectively, the combined properties preserve over a thousand acres of longleaf pine (LLP) flatwoods savannas. As of 1996 there were approximately 5,261 H of longleaf pine in St. Tammany Parish (Outcalt and Sheffield 1996). Land managers from both the Louisiana Department of Wildlife and Fisheries and The Nature Conservancy were aware of the threatened state of longleaf pine in southeast Louisiana when they proposed Lake Ramsay's primary management objective to 'restore, conserve and perpetuate all natural habitats and species indigenous to the area' (Louisiana Department of Wildlife and Fisheries 1992, The Nature Conservancy). In fact, one of the desired conditions was 'no or minimal exotic species'. These were low use areas with some hunting allowed, but most activity centered on research and education.

Because the area surrounding Lake Ramsay has had a history of wildfires the ground cover is rich in native species. There are over 100 plant species/0.40 H, many of which are only associated with longleaf pine flatwoods and their bogs (Louisiana Department of Wildlife and Fisheries 1992, The Nature Conservancy). However, most longleaf pines were clear-cut earlier in the century. Stands present on the property are from regeneration with few mature trees over 100 years old, and most between 20 and 80 years of age.

The soil profile at Lake Ramsay is a gradation due to streambed erosion. Soil characteristics are silt loam from Stough and Prentiss series on the raised terrain and silty clay loams from Myatt-Guyton series in depressions. Soils from these series have poor drainage qualities and are characteristic of flatwoods habitats in Louisiana (Amacher et al. 1989). William Patterson from the Department of Agronomy at Louisiana State University conducted soil analyses in 1996.

Lake Ramsay WMA is divided into fire blocks for management purposes. The boundaries follow both natural landscape and manmade features. Our research was conducted in the South Central 3 block (SC3) and the adjacent Nature Conservancy (TNC) property (Figure 1). Both sections of property are about 12.14 H each. SC3 is characterized as generally densely stocked longleaf pine forest mixed with open savanna with a good bit of longleaf pine (Louisiana Department of Wildlife and Fisheries 1992, The Nature Conservancy). Toothache grass, *Ctenium aromaticum*, and Muhley grass, *Muhlenbergia expansa*, are dominant grasses. Toothache grass is found in wetter areas and Muhley grass in drier areas. SC3 is bordered by bayhead forest (laurel oak, water oak, cyrilla woodland) on two sides and a road on the other. The TNC research site is typical longleaf pine savanna with characteristic clusters of longleaf pine in open, grassy fields. Private property on three sides and a road on the remaining side serve as boundaries for the site.

Plot Layout

The topography of the property allowed for establishment of six-0.81 H plots/site (Figure 2). From a center point, 8 flags marked the outer boundary of each circular plot. Another 8 flags marked an inner 0.20 H circle. Six pitfall trap pairs were arranged in

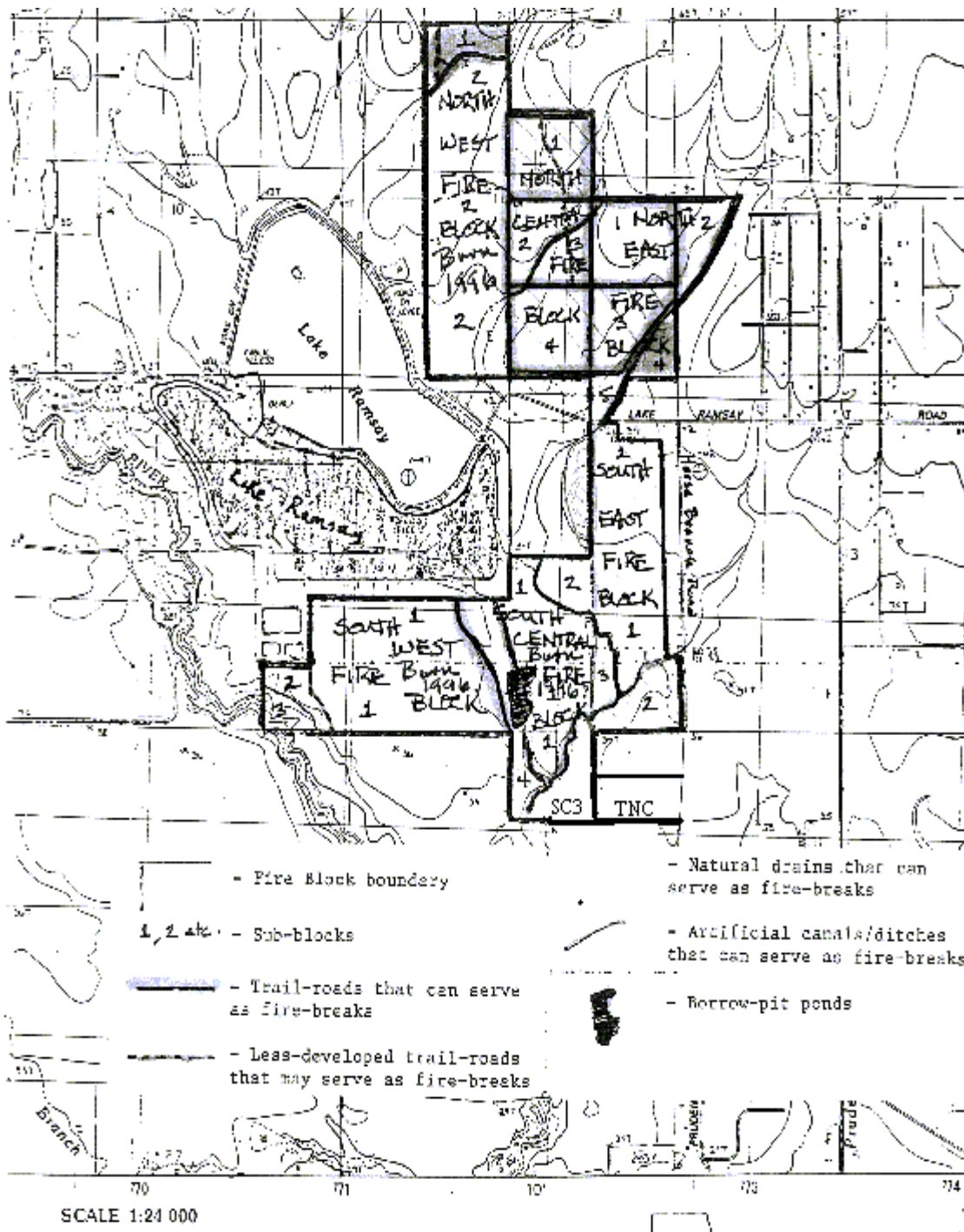


Figure 1: Map of Lake Ramsay Wildlife Management Area fire blocks. Research sites are located in the lower portion of the map, labeled SC3 and TNC.

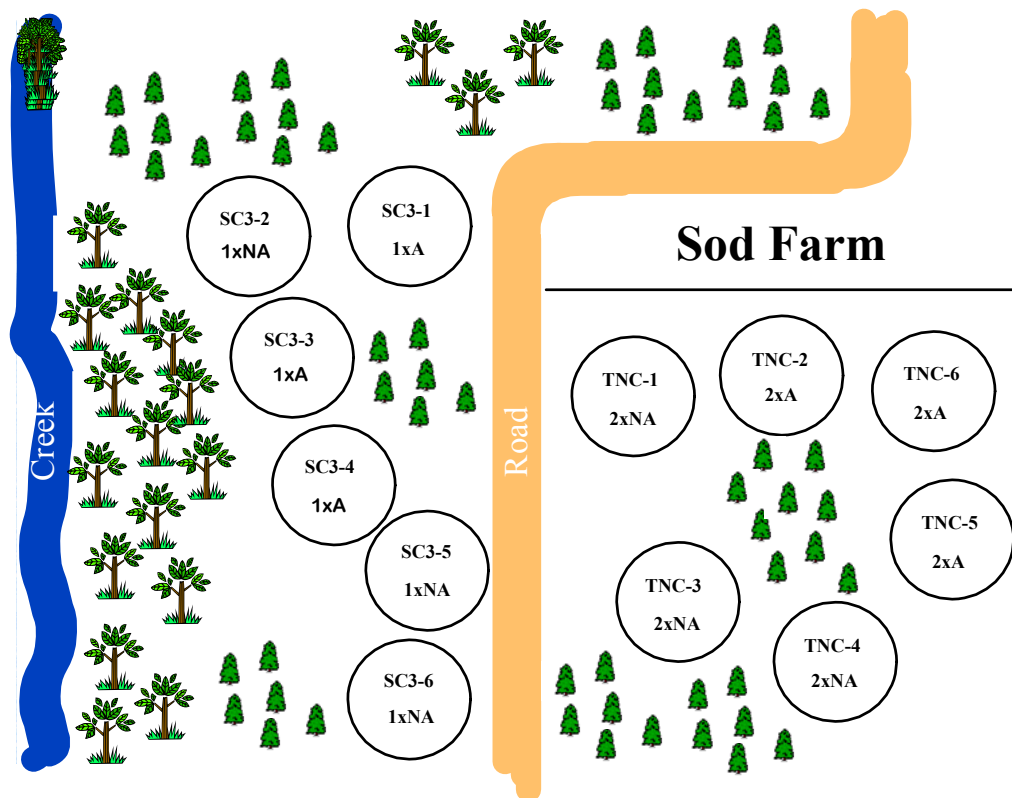


Figure 2: Plot and treatment arrangement within SC3 and TNC sites

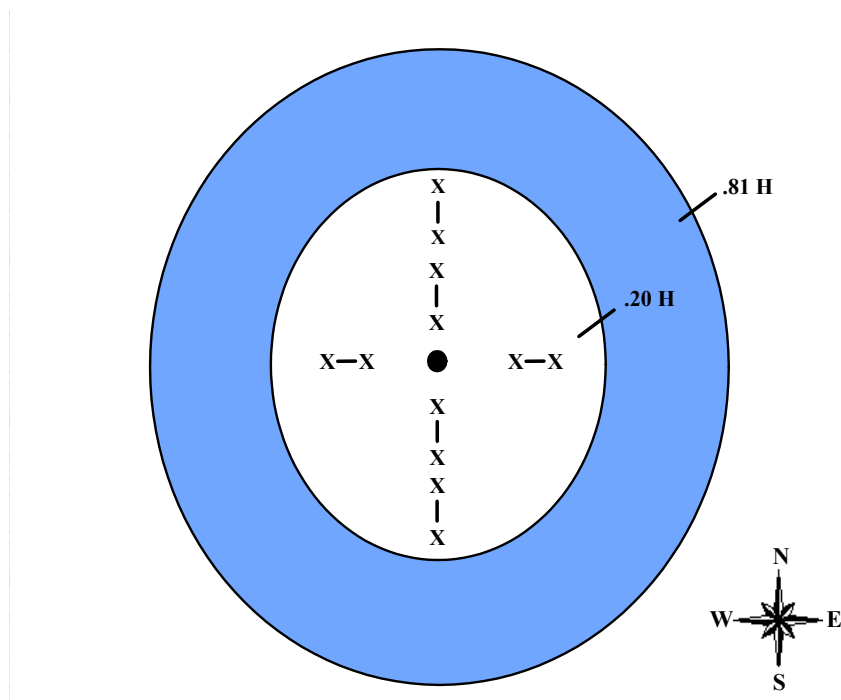


Figure 3: Pitfall trap arrangement within a plot.

north, south, east, west transects within the .20 H (Figure 3). Trap pairs were placed 10m apart.

Sampling Method

Pitfall trap pairs consisted of two 100ml, round centrifuge tubes (lipped) paired by a metal barrier. Sections of PVC pipe, 19.05 cm x 3.18 cm, were hammered into the soil to secure holes for tubes. Even though centrifuge tubes were lipped, a CREED #20 O-ring (3.02 cm x 2.54 cm x .24 cm) placed under the lip was necessary to keep the tube from sliding down into the PVC pipe. A no.3 egg fishing weight was placed in the bottom of each tube to keep the tube from floating in saturated soil. Each ensemble was covered by a 17.78 cm x 17.78 cm square of metal flashing held approximately 1.27 cm above tubes by two 15.24 cm galvanized common nails. The collecting preservative consisted of 3.79 L of Prestone LowTox™ mixed into 15.14 L 95% ethanol and 10 ml LiquiNox detergent. Approximately 40 mls of solution was poured into tubes during sampling. When not in use tubes were capped with #7 rubber stoppers. The stainless steel (24 gage) barriers were 91.44 cm x 10.16 cm with a 16.51 cm x 2.54 cm notch cut into each end making the barrier 'T' shaped. Barriers were inserted into the soil the depth of the notch.

Sampling was conducted twice per month for 48 hours each time. Samples were collected April through November. Sampling began during July 1996 and concluded during August 1999. Collections from year 1 consisted of samples from July 1996 through July 1997 (n=9), year 2 were from August 1997 through July 1998 (n=8), and year 3 were from August 1998 through August 1999 (n=9). Specimens were collected

and brought back to the lab for identification. Collections are housed in the Louisiana State Arthropod Museum at Louisiana State University.

Environmental Variables

Soil moisture was measured on a monthly basis during years 1 and 3. The equipment was unavailable for use during year 2. A Quickdraw 2900FI Soilmoisture Probe was used to measure soil water. The ceramic tip of the probe absorbed soil water; measurements were displayed as centibars. A reading of 0 indicated saturation. Measurements were made near the center of each plot within the first 5-10 cm of the soil surface.

Soil temperature was measured monthly using a Digital Longstem Thermometer. Measurements were taken near the center of each plot, 5 cm into the soil surface.

Leaf litter dry weight was recorded in plots during years 1 and 3. A 1m² frame was haphazardly tossed into each plot. Leaf litter was gathered using hand rakes and bagged in paper grocery bags. A flag was used to mark sample locations so that locations were not sampled twice. Bags were placed in a 37.78° C drying room for approximately two weeks. They were then weighed, dried 24 hrs, and reweighed to make sure litter had dried completely.

Treatments - Fire Frequency

The management plan for Lake Ramsay WMA proposed random fires every one to four years. However, at the time our study was initiated our research site had not undergone a managed fire. The last fire was thought to have taken place in 1988. Eight years without a fire in a LLP savanna meant that there was a substantial amount of fuel (pine needles, leaf litter) to be burned. The fuel load was too great to ignite a fire on each

plot for the first fire treatment. Rather, SC3 and TNC were each burned by a separate fire. TNC plots were randomly selected to receive two fires in a two-year period and plots on SC3 received a single fire. The first applied fire took place late in the growing season on August 19, 1997. The average leaf litter dry weight for plots was 591.73 g, ambient temperature ranged between 30.56 – 37.78 C, winds were 0 – 6.44 kph, and relative humidity was 42 – 64%. The second fire also took place late in the growing season on August 5, 1998. The fuel load was not uniform enough to light a single backfire on TNC plots; therefore, each plot was ignited in the center and allowed to burn out. Temperature ranged between 31.11 – 33.33 C, winds were 5.63 – 8.05 kph, and relative humidity was 54 – 62%.

Treatments - Amdro[®]

Amdro[®] a formicidal, granular bait was applied to reduce *S. invicta* activity on three randomly chosen plots per site. Amdro[®], the trade name for hydramethylnon, is in the Amidinohydrazone family of insecticides. Once ingested by ants, this toxicant interrupts the electron transport chain and stops metabolism. Amdro[®] was unstable in sunlight and broke down in about a day. The soybean oil attractant combined with a specific size granule attracted ‘big-headed’ ants to the bait. Ants at Lake Ramsay which could be categorized as ‘big-headed’ were *Crematogaster*, *Cyphomyrmex*, *Pheidole*, and *Solenopsis*. However, a study reported in 1984 concluded that with both broadcast and mound applications of Amdro[®], effect on native ants “appeared to be negligible”. Also, they did not detect bait residues after 24 h, and recommended repeated applications to maintain suppression of *S. invicta* (Apperson et al. 1984). Amdro[®] was applied at a rate of 5.44 kg/.81 H. This rate was obtained by applying the bait at the

recommended amount of 73.93 ml directly around *S. invicta* mounds. I applied bait around mounds rather than broadcasting to minimize exposure of other insects to the bait. Amdro[®] was applied on the following dates: 10/4/97, 5/5/98, 8/18/98, and 5/24/99. All field procedures that focused on *S. invicta* were conducted during times of the day when these ants were most likely to be active or near the surface of mounds. Researchers have recommended times when temperatures were between 21 – 32° C.

We monitored *S. invicta* activity after Amdro[®] applications by checking mounds for brood, alates, and workers and by placing peanut butter and honey baited vials within the .20 H circle of each plot. Approximately two weeks following Amdro[®] applications we checked for activity within each mound by excavating the center of each mound. If alates, brood, and/or greater than 100 workers were observed mounds in that portion of the plot were retreated the following week.

Vials baited with peanut butter or honey were alternately placed 10 m apart along the same transects as pitfall traps. Starting with a peanut butter vial in the center of the plot, 5 hinged cap vials (60 ml) were placed along each transect. Vials were left open for 1 h before being closed and frozen. Ants were then counted and identified. This procedure was conducted once per month. When *S. invicta* activity appeared to be reaching levels observed on untreated plots, Amdro[®] was reapplied.

Because *S. invicta* abundance was a key factor in this research, knowing if colonies were monogyne or polygyne was important. We determined them as monogynous by excavating ants from the centers of mounds and sprinkling them over a piece of 1.22 m x 1.22 m white poster board. When a queen was observed, the search for other queens from the same mound continued until the surveyor was satisfied the colony

was monogynous. Excavations were done following a rain shower because ants are likely to be closer to the surface. Members from the USDA, APHIS, PPQ Imported Fire Ant Laboratory, Gulfport, MS verified our findings (personal communication, Homer Collins).

Treatments - Summary

Our treatments were grouped as follows: One applied fire and Amdro[®] (1xA), one applied fire and no Amdro[®] (1xNA), two applied fires and Amdro[®] (2xA), and two applied fires and no Amdro[®] (2xNA) (Figure 2).

	Amdro [®]	No Amdro [®]
1 fire	1xA	1xNA
2 fires	2xA	2xNA

Treatment assignment to plots within each site was random. However, all plots within a site received the same fire frequency. This study was arranged as a split plot design, but only sub-plots factors were replicated. Treatments listed above are the interaction of fire frequency (whole-plot) and Amdro[®] applications (sub-plot). Because whole-plot factors were not replicated only the interaction of whole-plots with sub-plots could be analyzed.

Analysis - Pseudoreplication

Roughly 50% of all research is nonreplicated and is either analyzed incorrectly or not analyzed at all (Miliken and Johnson 1989). Studies that include managed fires within the treatment structure are pseudoreplicated because, “it is impossible to replicate the disturbance event” (Van Mantgem et al. 2001). Managed applied fires, like the ones in this study, are not randomized or replicated. When considering management

implications, accurate assessments of species responses to treatments with applied fire are important. Despite these problems, four components of my design allowed for greater reliability in my results: (1) inferences from results pertained only to study sites, (2) multiple plots were established, (3) pretreatment sampling was conducted, and (4) measurements were repeated in time. Subsampling in place of true replicates can produce a reduced estimate of the error and increase the possibility of committing a Type 1 error (Hurlburt 1984). However, by establishing multiple plots within a site, I “strengthened my ability to accurately assess fire responses” (Van Mantgem et al. 2001). Time series designs inherently suffer from temporal autocorrelation. As I explain in the following section, a heterogeneous autoregressive repeated measure analysis of variance was used to analyze the data. This particular variance/covariance matrix structure best fit the data for reasons explained below. Collection of pretreatment data also enhanced interpretation of changes within the response variables.

Analysis - Repeated Measures Analysis of Variance

Repeated measures analysis of variance was used to calculate treatment differences on native and exotic ant abundance. Pretreatment data were included as a covariate to reconcile inherent plot differences that may have influenced results. Adjustments of the variance using an overdispersion parameter, such as SAS’s glimmix macro, prevented ‘inflated test statistics’ (Littell et al. 1996). Count data have been considered a Poisson distribution, but recent work has suggested such data were overdispersed; i.e., the variance was larger than the mean rather than equal to it. The heterogeneous autoregressive model Type 1 gave the desired extra-dispersion scale equal to one. This particular model allowed for variance to change with time.

Analysis - Rank Abundance

Rank abundance plots were used to present species abundance data for ants (Magurran 1988). Pretreatment samples were plotted separately from post-treatment samples for comparison. Because *S. invicta* was many orders more abundant than other species, it was excluded from graphs to maintain a readable scale for all other species.

Species dominance was graphed as a log abundance of species regressed against that species' rank. Lines were generated and treatments compared using Proc GLM regression analysis (SAS Institute, Inc. 2001). *Solenopsis invicta* was suspected to be an outlier in analyses due to its overwhelming abundance. This was confirmed by plotting residuals. As a result, analyses were done with and without *S. invicta*.

Analysis - Summary of Ant Species: Amdro[®] and Fire

A list of ant species collected was compiled along with raw counts of species by treatments and presented in table form. Pretreatment data (year 1) was listed separately from post-treatment data (years 2–3). Discussion of species in reference to impacts of Amdro[®] and fire applications were recounted separately.

Analysis - Sorenson Index

Sorenson's index was used to measure β diversity between treatments. This index measures the similarity between treatments by the equation $C_s = 2j/(a+b) \times 100$ where j is the number of species found in both treatments and a is the number of species found in treatment A and b the number of species found in treatment B (Magurran 1988). Complete similarity equals 100%. I used this index to compare all ant species and native ant species both pre (year 1) and post-treatment application (years 2-3).

Analysis - Diversity Indices

To better understand changes that occurred between each of the study years, I calculated four measures of diversity: numbers of individuals, species richness, evenness, and Shannon diversity for each treatment year. Species richness is the number of species collected, Shannon diversity, $H' = -\sum p_i \log p_i$ where p_i is the proportion of individuals in the i th species, is a “logarithmic measurement of the diversity of species weighted by the relative abundance of each species”, and evenness, $E = H' / \ln \text{richness}$, is a measure of how equally abundant species were (McCune and Mefford 1999, Ricklefs 1990). PC-ORD, a statistics package for multivariate analysis of ecological data, was used to generate diversity indices. Raw data for native and exotic ants (excluding *S. invicta*) were imported into PC-ORD using a compact data format (McCune and Mefford 1999). Indices were calculated on data pooled from all three replicates within each of the four treatments, allowing collection dates within study years to be treated as replicates. Statistics for each index were then generated using Proc Mixed ANOVA (SAS Institute, Inc. 2001). Least squares means and standard errors were presented.

Analysis - Beetles

Beetles from the families Carabidae and some Staphylinidae were identified from pitfall trap collections. Small sample sizes dictated that we use Fisher’s Chi-Square test for equal proportions to compare pre- (year 1) and post- (year 2) treatment collections and a Chi-square test of homogeneous proportions was used to analyze treatment differences from years 2 and 3.

Analysis - Environmental Parameters

All environmental parameters were analyzed using a repeated measures analysis of variance (Proc Mixed, SAS Institute 2001). The models were repeated on time. Soil temperatures and leaf litter dry weights were included in repeated measures analyses to determine if either contributed to changes in ant abundances within the treatment structure.

RESULTS

Repeated Measures ANOVA

Abundances of ants did not differ among plots prior to experimental treatments. Abundance of neither *S. invicta* ($F = 0.39$, $df = 1, 10$, $P = 0.54$) nor other species combined ($F = 3.41$, $df = 1, 10$, $P = 0.09$) were significantly different. Fire frequency and Amdro[®] had significant impacts on *S. invicta* abundance (Table 1). *Solenopsis invicta* was more abundant on 2xNA plots than in any other treatment (Table 2). Treatments that included Amdro[®] had fewer *S. invicta* than those without Amdro[®]. No differences in relative abundance of *S. invicta* were detected between 2xA and 1xNA, which was relevant from a management perspective. Significant date and treatment*date factor effects were most likely due to application of treatments being confounded with specific dates. The data set for other exotic ants was too small to analyze using the same statistics as applied to *S. invicta* and native species.

Table 1: Source table for *S. invicta*.

Source	NDF	DDF	Type III F	Pr > F
Treatment	3	8	31.22	< .0001
Date	16	128	12.38	< .0001
Treatment * Date	48	128	1.69	0.0110

Table 2: Least square mean differences for significant treatment effects on *S. invicta*. Adjusted P values represent a Tukey-Kramer adjustment.

Treatment	Estimate	Std Error	DF	T	Adj P
2xA 2xNA	-1.1871	0.1959	8	-6.06	0.0014
2xA 1xA	0.9500	0.3429	8	2.77	0.0919
2xA 1xNA	-0.5180	0.2180	8	-2.38	0.1597
2xNA 1xA	2.1371	0.2968	8	7.20	0.0004
2xNA 1xNA	0.6691	0.1343	8	4.98	0.0047
1xA 1xNA	-1.4690	0.3118	8	-4.71	0.0066

No combination of fire and Amdro[®] had a significant impact on native ant relative abundance (Table 3).

Table 3: Source table for native ants.

Source	NDF	DDF	Type III F	Pr > F
Treatment	3	8	2.35	0.1488
Date	16	128	8.62	< .0001
Treatment * Date	48	128	0.95	0.5715

Rank Abundance

Two species (excluding *S. invicta*) were dominant in collections prior to treatment applications were *Paratrechina faisonensis* and *Crematogaster pilosa*, both native (Figure 4). Following treatments, dominance shifted to two exotics, *Cyphomyrmex rimosus* and *Pheidole flavens* (Figure 5). Statistical comparisons of rank abundance among the four treatments with and without inclusion of *S. invicta* indicated no significant differences ($F = 1.21$ $df = 60, 16$, $P = 0.35$; $F = 1.24$ $df = 57, 16$ $P = 0.33$) (Figures 6 & 7). In other words, no treatment was different from any other with respect to order of species' abundance.

Summary of Ant Species - Fire

There were 26 species of ants collected from pitfall traps from August 1996 through August 1999. Exotic species comprised 23% of species collected and 98% of the number of individuals collected from pitfall traps. *Solenopsis invicta* alone accounted for 95% of individuals trapped (Table 4 in bold). When *S. invicta* data were excluded the remaining exotics and natives were relatively equal in abundance with a ratio of 53 percent. Three additional species were collected from peanut butter and honey baited vials: *Crematogaster missouriensis*, *Pseudomyrmex pallidus*, and *Tapinoma sessile*.

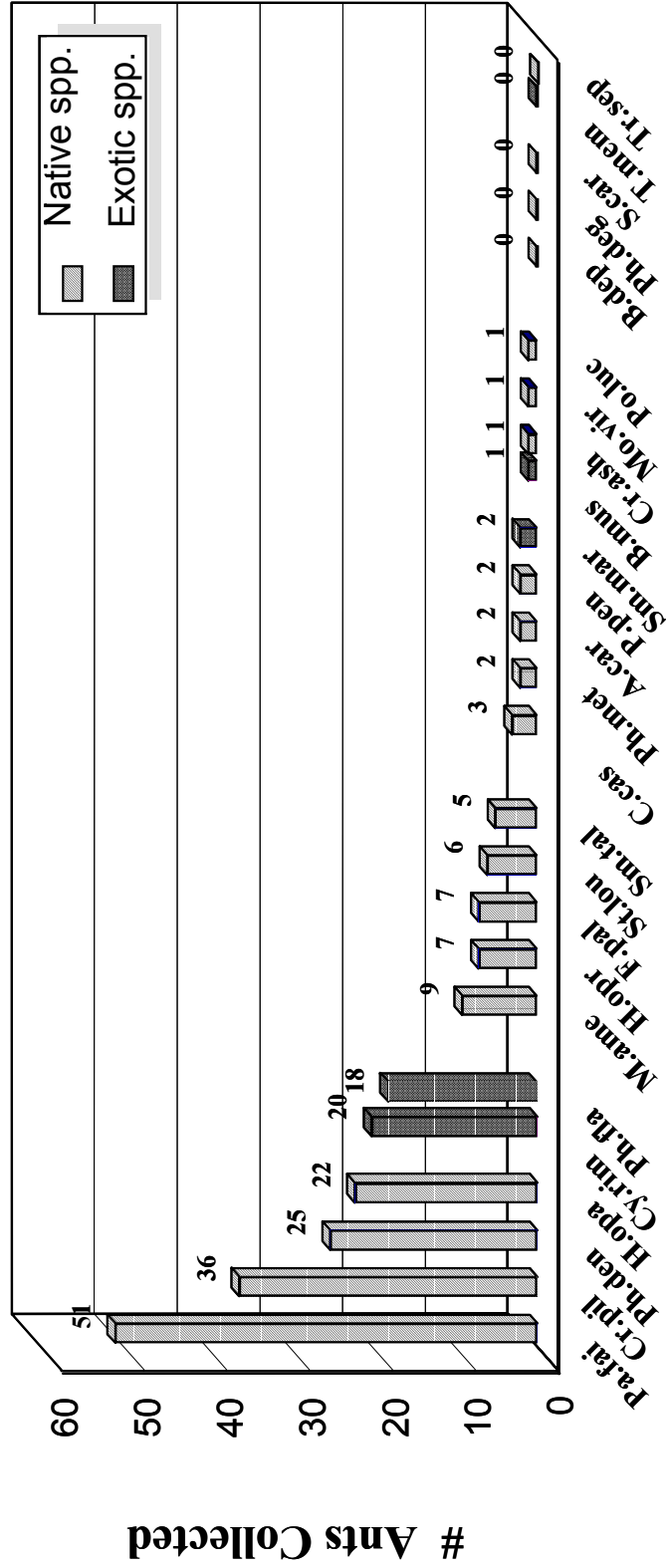
The application of fire tended to increase abundance and richness of all ants (Table 4, Before and After). Exotic species such as: *Brachymyrmex musculus*, *Cyphomyrmex rimosus*, *Pheidole flavens*, and *S. invicta* increased in abundance in all treatments following fire. The same was true for native species: *Crematogaster pilosa*, *Formica pallidefulva*, *Hypoponera opaciceps*, *Paratrechina faisonensis*, and *Strumigenys louisianae*. Despite pooling of years 2 and 3 in the “After” column increases in abundance of exotic and native species were dramatic and much greater than twice as much. However, most native species were not altered by fire.

Treatment 2xNA had the greatest percent increase in exotic abundance (5%) and the greatest drop in native abundance (5%), but ended up with the most native abundance overall (287). The opposite was true for treatment 1xA which had a 3% decrease in exotic abundance and a 3% increase in native abundance. This treatment group was second in native ant abundance with 190 individuals, a 5-fold increase from initial counts. Despite having the least number of native ants collected, treatment 1xNA had the greatest increase in native abundance, 8 times the initial count.

Descriptive comparisons of species richness differences revealed 1xNA and 2xA had the same and lowest ratio of exotics to natives (5:14). Applying fire once, then allowing plots to regenerate for a year resulted in fewer exotics and more native species in the savanna. The same results were obtained by applying fire to plots two years in a row and applying Amdro[®] during those years (Table 4).

Summary of Ant Species - Amdro[®]

Species richness increased more in treatments without Amdro[®] than treatments with Amdro[®] (Table 4 Before and After). Looking at the impact of Amdro[®] on species’



Ant Species

Figure 4: Rank abundance plot of ant species collected from pitfall traps during year 1. *Solenopsis invicta* (n = 3,693) was removed to maintain graph scale for species presented. Data were pooled across treatments.

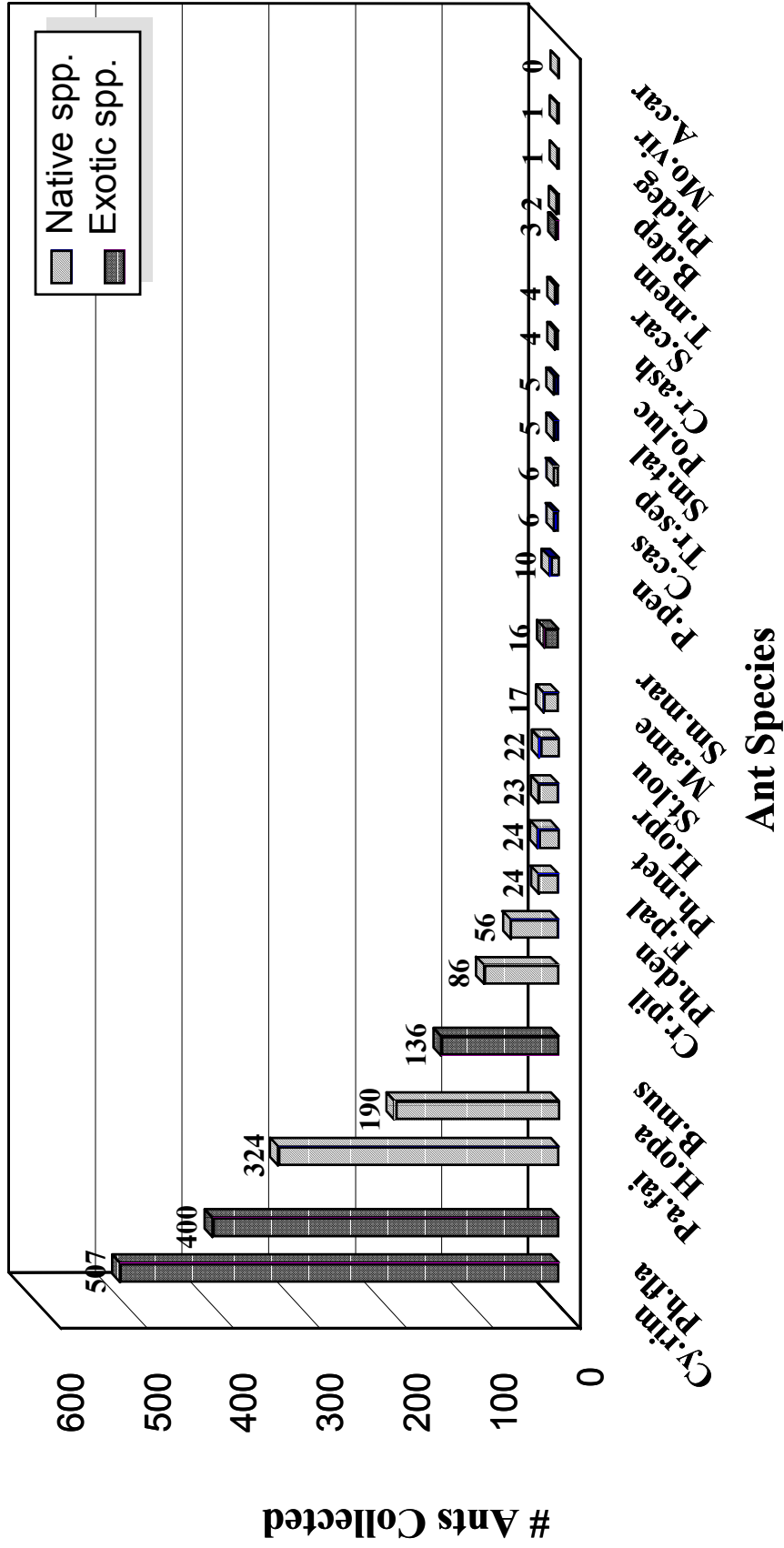


Figure 5: Rank abundance plot of ant species collected from pitfall traps during years 2-3. *Solenopsis invicta* (n = 36,490) is removed to maintain scale for species presented. Data were pooled across treatments.

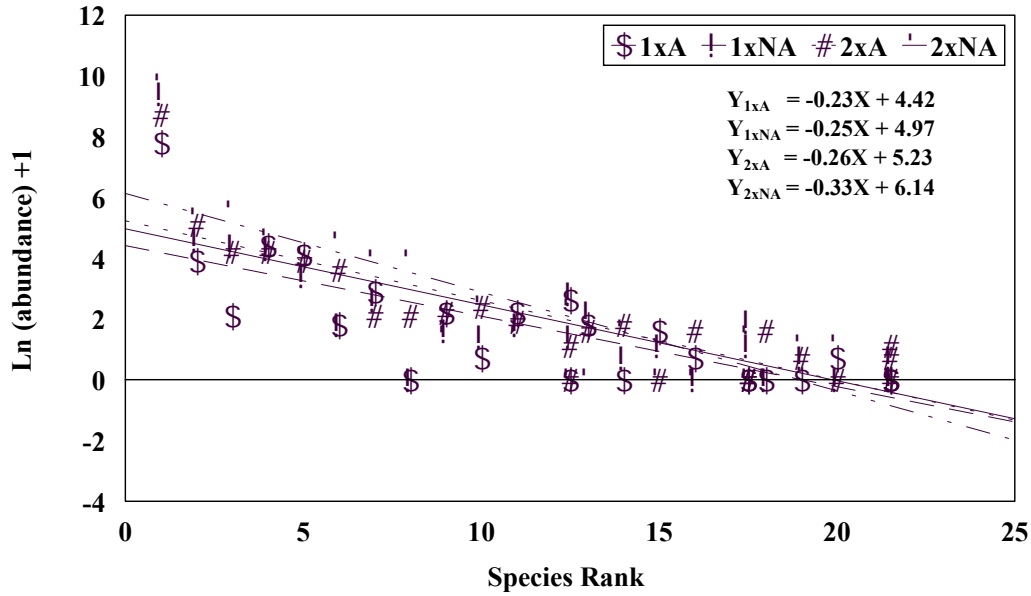


Figure 6: Regression analysis of all species' ranks and abundances within the four treatments. Ant species were collected from pitfall traps during years 2-3.

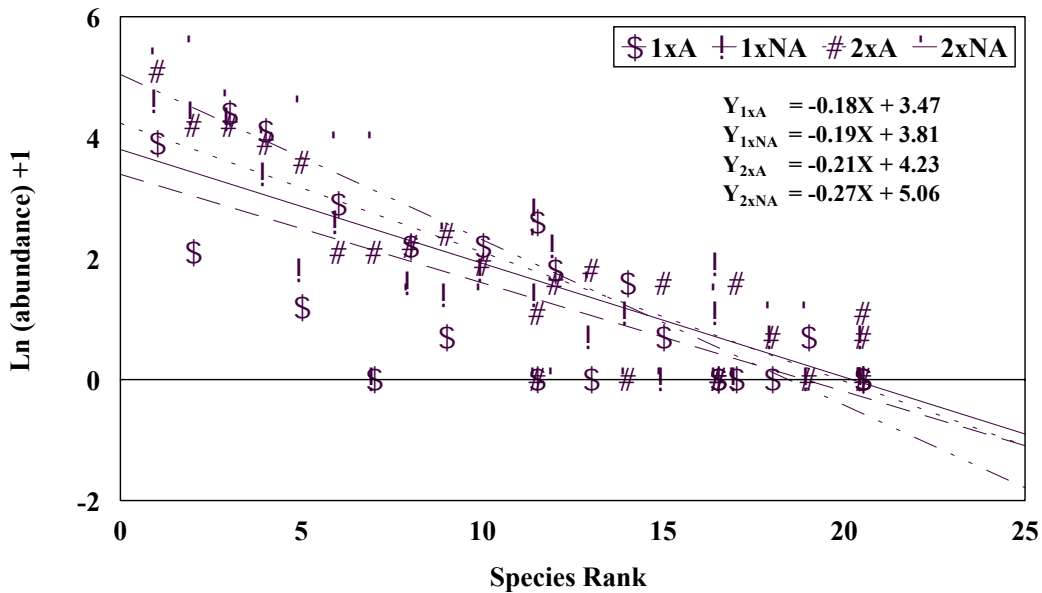


Figure 7: Regression analysis of species' ranks and abundances (excluding *S. invicta*) within the four treatments. Ant species were collected from pitfall traps during years 2-3.

Table 4: Summary of ant species collected from pitfall traps for each treatment. Totals in ‘Before’ columns are data from year 1 (n=9) collected prior to treatment applications. Totals in ‘After’ columns are data from years 2-3 (n=17) compiled after all treatments were applied. Species in bold are exotic.

* *S. invicta* excluded

Species	Treatment							
	1xA		1xNA		2xA		2xNA	
	Before	After	Before	After	Before	After	Before	After
<i>Aphaenogaster carolinensis</i>	0	0	0	0	2	0	0	0
<i>Brachymyrmex depilis</i>	0	0	0	0	0	2	0	0
<i>Brachymyrmex musculus</i>	0	5	0	5	1	34	0	92
<i>Camponotus castanea</i>	1	4	2	2	0	0	0	0
<i>Crematogaster ashmeadi</i>	0	0	0	0	1	4	0	0
<i>Crematogaster pilosa</i>	2	18	4	12	1	7	29	49
<i>Cyphomyrmex rimosus</i>	4	48	8	102	6	159	2	198
<i>Formica pallidefulva</i>	0	1	1	3	6	10	0	10
<i>Hyponera opaciceps</i>	6	60	6	29	6	48	4	53
<i>Hyponera opacior</i>	1	8	0	4	3	8	3	3
<i>Monomorium viride</i>	0	0	1	1	0	0	0	0
<i>Myrmecina americana</i>	4	12	0	3	2	2	3	0
<i>Paratrechina faisonensis</i>	13	78	2	76	24	68	12	102
<i>Pheidole dentata</i>	0	0	1	0	21	7	3	49
<i>Pheidole dentigula</i>	0	0	0	0	0	1	0	0
<i>Pheidole flavens</i>	0	7	1	80	17	64	0	249
<i>Pheidole metallescens</i>	0	0	2	15	0	0	0	9
<i>Polygerus lucidus</i>	1	0	0	2	0	0	0	3
<i>Ponera pennsylvanica</i>	0	0	1	1	1	5	0	4
<i>Smithistruma margaritae</i>	1	5	0	8	1	4	0	0
<i>Smithistruma talpa</i>	1	1	0	0	4	4	0	0
<i>Solenopsis carolinensis</i>	0	0	0	1	0	1	0	2
<i>Solenopsis invicta</i>	808	2,553	879	11,560	1,232	6,176	774	16,201
<i>Strumigenys louisianae</i>	4	8	0	5	1	6	1	3
<i>Trachymyrmex septentrionalis</i>	0	0	0	6	0	0	0	0
<i>Trichoscapa membranifera</i>	0	1	0	0	0	0	0	2
Total # Individuals	846	2,809	908	11,915	1,329	6,610	831	17,029
# Native Individuals	33	197	19	159	72	173	55	469
# Exotic Individuals *	5	59	10	196	25	261	2	539
Native Species Richness	9	11	9	14	12	15	7	11
Exotic Species Richness	3	4	3	5	5	4	2	5

abundances within fire treatments indicated that receiving a single fire and Amdro[®] reduced *S. invicta* abundance by 4.5 times (11,560:2,553), but receiving two fires and Amdro[®] only reduced *S. invicta* abundance by 2.6 times (16,201:6,176). Amdro[®] had greater efficacy in the absence of repeated fire disturbance. The same can be said for reduction of other exotics as well. Other exotic species' abundances declined 3-fold on single fire plots (195:66) and 2-fold on plots receiving two fires (541:261) (Table 4).

Comparisons between fire frequencies over the duration of the study (Table 4 After) indicated a 2.5-fold increase in *S. invicta* abundance between 1xA and 2xA treatments (2,553:6,176) compared to a 1.4-fold increase between 1xNA and 2xNA treatments (11,560:16,201). A greater increase in *S. invicta* abundance on Amdro[®] treated plots indicated suppression efforts caused a greater proportional increase in *S. invicta* numbers overall. In other words, when *S. invicta* was suppressed by Amdro[®] the increase due to multiple fires was greater than when *S. invicta* was not suppressed. Even though the gain in *S. invicta* was greater on Amdro[®] treated plots, those plots still had 50 – 75% fewer *S. invicta* than plots without Amdro[®]. This trend was also true for other exotics. Amdro[®] treated plots experienced a 4-fold difference (66:261) in exotic species' abundances compared to plots without Amdro[®] which had a 3-fold difference (195:541) (Table 4).

Diversity Indices

In general, there was greater similarity of ant species between treatments following the initial fire (Table 5 and 6). That is, species composition from one treatment to another was more similar after year 1. Most of the increase in similarity can be

attributed to addition of exotic species. For example, *T. membranifera* was not collected prior to treatment application and then appeared in treatments 1xA and 2xNA only after treatments were applied. Also, *B. musculus* and *P. flavens* were not present in all treatments initially, but were eventually collected from all treatments (Table 4).

Table 5: Sorenson’s Index of Similarity for all ant species collected from pitfall traps. ‘Before’ represents data collected prior to treatment application, year 1. ‘After’ represents data collected after all treatments were applied, years 2-3.

Treatment	1xA		1xNA		2xA	
	Before	After	Before	After	Before	After
1xA	-	-				
1xNA	50%	76%	-	-		
2xA	64%	76%	67%	74%	-	-
2xNA	80%	65%	55%	80%	69%	74%

Table 6: Sorenson’s Index of Similarity for native ant species collected from pitfall traps. ‘Before’ represents data collected prior to treatment application, year 1. ‘After’ represents data collected after all treatments were applied, years 2-3.

Treatment	1xA		1xNA		2xA	
	Before	After	Before	After	Before	After
1xA	-	-				
1xNA	44%	64%	-	-		
2xA	60%	64%	64%	64%	-	-
2xNA	80%	55%	47%	80%	74%	72%

Appearance of native species following treatments was not as influential as that of exotics. However, two native species did contribute to increased similarity of species. *S. carolinensis* was not collected prior to treatments and *F. pallidefulva* increased its distribution to include all treatment plots following treatment application (Table 4). There was a decrease in similarity between 1xA and 2xNA even though both treatments gained 3 exotic species: treatment 2xNA gained 5 native species but 1xA only gained 1

(Table 4, 5, and 6). The difference of 4 native species resulted in a reduction in similarity between the treatments.

Numbers of native ants generally increased due to the first applied fire and then dropped back to pretreatment levels by year 3 (Table 7, Figure 8). Numbers of native species increased significantly after the first fire and only decreased in year 3 on plots that received a single applied fire (Table 7, Figure 8). Treatment 2xA was the exception with a slight gain in abundance and richness whereas diversity remained unchanged (Table 7, Figure 8).

Abundance of exotics (excluding *S. invicta*) in all treatments increased from year 1 to 2 following the first applied fire (Table 8, Figure 9). They exhibited apparent differences in abundance and richness in response to fire frequencies from year 2 to 3 regardless of Amdro[®] applications. Treatments that included two fires generally had a continuous increase in abundance and richness of exotics from year 1 to 3. In contrast, exotics from single fire treatments increased in abundance and richness following the first applied fire and then decreased in year 3.

Beetles

The number of beetles collected significantly increased following the first applied fire ($X^2_1 = 12.16$ $p = 0.00$). Numbers of beetles more than doubled from year 1 to year 2 going from 24 to 55. Only 8 of 24 carabid species and 2 of 4 identified staphylinid species were collected prior to the first applied fire (Table 9). There were approximately 15 additional Staphylinid species not identified and therefore not included in the analyses. The Chi-square test of homogeneous proportions for beetles collected from the

Table 7: Comparison of diversity indices for native ant species from pitfall trap data: all years. Least squares means and standard errors for each index are shown.

^ans = not significant

Treatment	Number of Individuals	Species Richness	Evenness	Shannon Diversity
2xNA: year 1	9.67 (4.35)	3.50 (.73)	.67 (.12)	.84 (.18)
2xNA: year 2	23.13 (3.77)	4.75 (.63)	.80 (.11)	1.12 (.16)
2xNA: year 3	11.88 (3.77)	3.50 (.63)	.82 (.11)	.95 (.16)
2xA: year 1	8.25 (3.77)	3.63 (.63)	.79 (.11)	1.09 (.16)
2xA: year 2	8.25 (3.77)	3.88 (.63)	.78 (.11)	1.08 (.16)
2xA: year 3	12.13 (3.77)	4.00 (.63)	.67 (.11)	.94 (.16)
1xNA: year 1	3.40 (4.76)	3.00 (.80)	.78 (.13)	.97 (.20)
1xNA: year 2	15.57 (4.01)	5.43 (.68)	.78 (.11)	1.27 (.17)
1xNA: year 3	4.88 (3.77)	2.13 (.63)	.46 (.11)	.43 (.16)
1xA: year 1	5.50 (4.35)	2.83 (.73)	.71 (.12)	.83 (.18)
1xA: year 2	17.63 (3.77)	3.88 (.63)	.77 (.11)	.98 (.16)
1xA: year 3	4.75 (3.77)	1.88 (.63)	.58 (.11)	.45 (.16)
F Test	P = 0.0162	P = 0.0154	ns ^a	P = .0112

Table 8: Comparison of diversity indices for exotic ant species (excluding *S. invicta*) from pitfall trap data: all years. Least squares means and standard errors for each index are shown.

Treatment	Number of Individuals	Species Richness	Evenness	Shannon Diversity
2xNA: year 1	2.00 (10.38)	1.00 (.53)	0 (.33)	0 (.33)
2xNA: year 2	26.00 (3.67)	2.38 (.27)	.59 (.12)	.54 (.12)
2xNA: year 3	36.50 (3.46)	2.88 (.27)	.87 (.11)	.90 (.11)
2xA: year 1	4.17 (4.24)	2.00 (.31)	.73 (.13)	.57 (.13)
2xA: year 2	10.63 (3.67)	2.00 (.27)	.38 (.12)	.38 (.12)
2xA: year 3	15.25 (3.46)	2.38 (.25)	.59 (.11)	.55 (.11)
1xNA: year 1	4.50 (7.34)	1.50 (.53)	.36 (.24)	.25 (.23)
1xNA: year 2	17.13 (3.67)	2.13 (.27)	.47 (.12)	.42 (.12)
1xNA: year 3	7.25 (3.67)	2.00 (.27)	.67 (.12)	.55 (.12)
1xA: year 1	2.50 (7.34)	1.50 (.53)	.41 (.23)	.28 (.23)
1xA: year 2	5.71 (3.92)	2.29 (.29)	.85 (.12)	.70 (.12)
1xA: year 3	2.38 (3.46)	1.13 (.25)	.08 (.11)	.06 (.11)
F Test	P < 0	P = 0.02	P < 0	P < 0

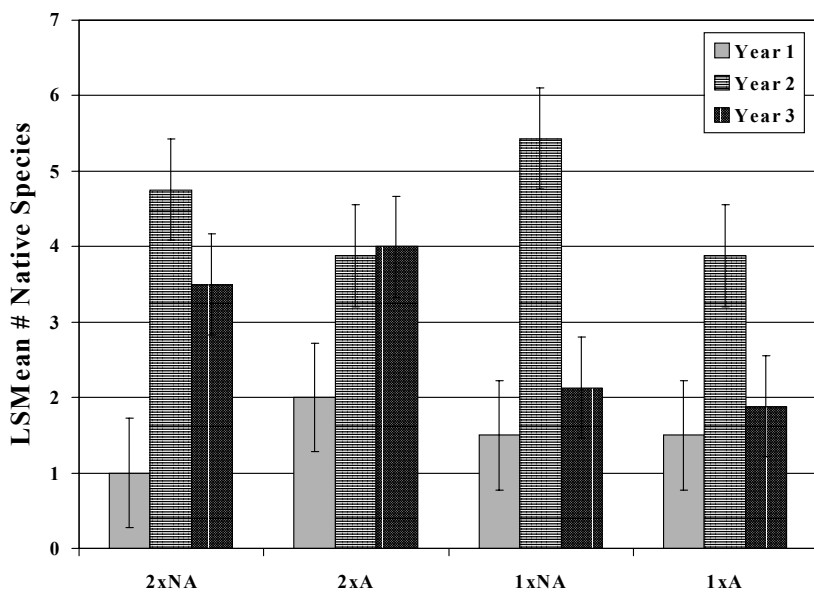
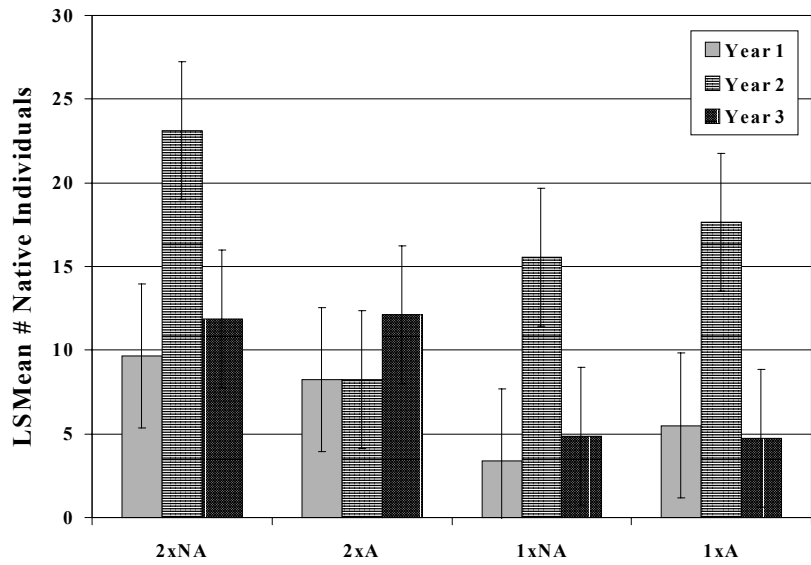


Figure 8: Least squares means for native species abundance (top) and richness (bottom): all years.

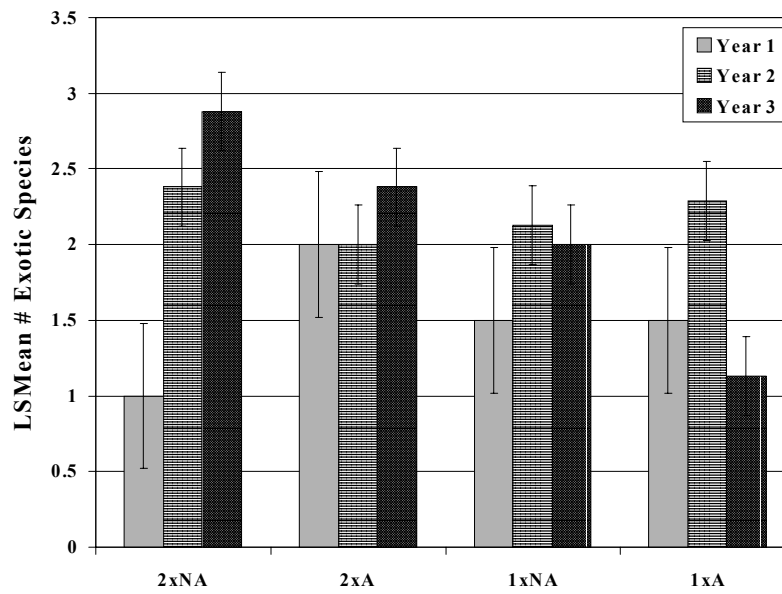
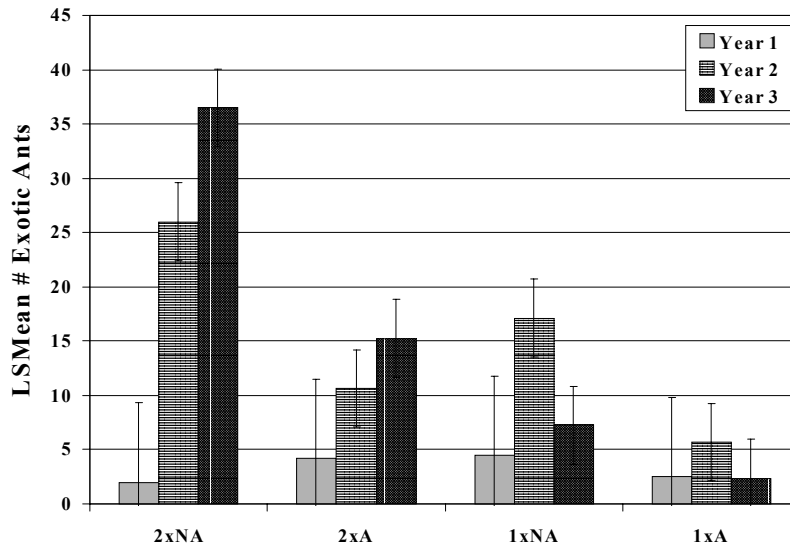


Figure 9: Least squares means for exotic species (excluding *S. invicta*) abundance (top) and richness (bottom): all years.

four treatments indicated the distribution of values were not equal ($X^2_1 = 7.51$ $p = 0.01$). Chi-square cell frequencies indicated treatment 1xNA had half the number of beetles collected compared to the other three treatments (Table 10).

Environmental Parameters

There were no differences in soil moisture, leaf litter dry weights or soil temperatures among plots prior to treatment applications (moisture: $F_{1,10} = 0.12$ $P = 0.73$, litter: $F_{1,10} = 1.93$ $P = 0.19$, temperature: $F_{1,4} = 1.75$ $P = 0.26$). Mean soil moisture was at 97% saturation during the first sampling season. Post treatment soil moisture comparisons again showed no difference among treatment plots ($F_{1,10} = 2.10$ $P = 0.18$). Soil moisture remained high with a mean 96% saturation. Soil remained saturated for most of the study except August, which was the driest month (Figure 10).

Soil temperatures were significantly different among fire treatments, but only in year 3 ($F_{1,4} = 24.32$ $P = 0.01$). Plots burned twice were approximately 2° C warmer than plots receiving a single applied fire. Least squares means estimates were 25.46 and 23.77 respectively. Soil temperatures did vary significantly by date during all three years (Figure 11). Soil temperatures did not contribute to differences in ant abundance due to treatments ($F = 3.36$ $df = 1, 7$ $P = 0.11$).

Leaf litter dry weights were significantly different among treatments ($F_{1,10} = 26.44$ $P = 0.00$) with plots that received two fires having half the amount of litter (140.40 g) as plots that received only one applied fire (282.01 g). Treatment differences in ant abundance were not influenced by changes in leaf litter ($F_{1,1} = 2.49$ $P = 0.36$). On plots that received a single applied fire, leaf litter reached 60% (282:466 g) of preburn amounts by the study's end.

Table 9: Summary of identified beetle species collected from pitfall traps for each treatment. Totals in ‘Before’ columns were data from year 1 (n=9) collected prior to treatment applications. Totals in ‘After’ columns were data from years 2-3 (n=17) compiled after all treatments were applied.

Species name	Treatment							
	1xA		1xNA		2xA		2xNA	
	Before	After	Before	After	Before	After	Before	After
Carabidae								
<i>Acupalpus</i> sp.						1		
<i>Agonum</i> sp.								1
<i>Aspidoglossa subangulata</i>		2		1		4		
<i>Badister reflexus</i>		1						
<i>Calosoma alternans</i>								1
<i>Calybe sallei</i>		1		3		1		
<i>Chlaenius (Anomoglossus) amoenus</i>	1						2	
<i>Elaphropus</i> sp.	6	1		1	4	3	3	5
<i>Helluomorphoides nigripennis</i>		1				1		1
<i>Loxandrus crenatus</i>						1		
<i>Loxandrus pravitubus</i>				1	1			
<i>Mioptachys</i> sp.								1
<i>Oodes americanus</i>			1	1				1
<i>Oxydrepanus rufus</i>								1
<i>Poecilus</i> sp.		1						
<i>Pseudaptinus lecontei</i>					1	1		1
<i>Pterostichus (Piemus) monedulus</i>		1						
<i>Selonophorus</i> sp. 1				1				1
<i>Selonophorus</i> sp. 2				2		1		
<i>Semiardistomis puncticollis</i>		8	1	1	1	9		10
<i>Semiardistomis veridis</i>		4						
<i>Tachys</i> sp. 1	1	1		1		2		
<i>Tachys</i> sp. 2		2				1		2
<i>Trechus</i> sp.					1			
Staphylinidae								
<i>Arthmius</i> n. sp.		8				1		6
<i>Myrmecosanus ferrugineus</i>								1
<i>Pselaphus</i> sp.	1	1						
<i>Reichenbachia louisiana</i>		1				2		
Total # Individuals	8	33	1	12	8	28	5	32

Table 10: Chi-square table of fire frequency by Amdro[®] for beetles collected during years 2-3.

FIRE Frequency Percent Row Percent Column Percent	AMDRO		
	Amdro [®]	No Amdro [®]	TOTAL
1x	33	12	60
	31.43	11.43	57.14
	73.33	26.67	
	54.10	27.27	
2x	28	32	45
	26.67	30.48	42.86
	46.67	53.33	
	45.90	72.73	
TOTAL	61	44	105
	58.10	41.90	100.00

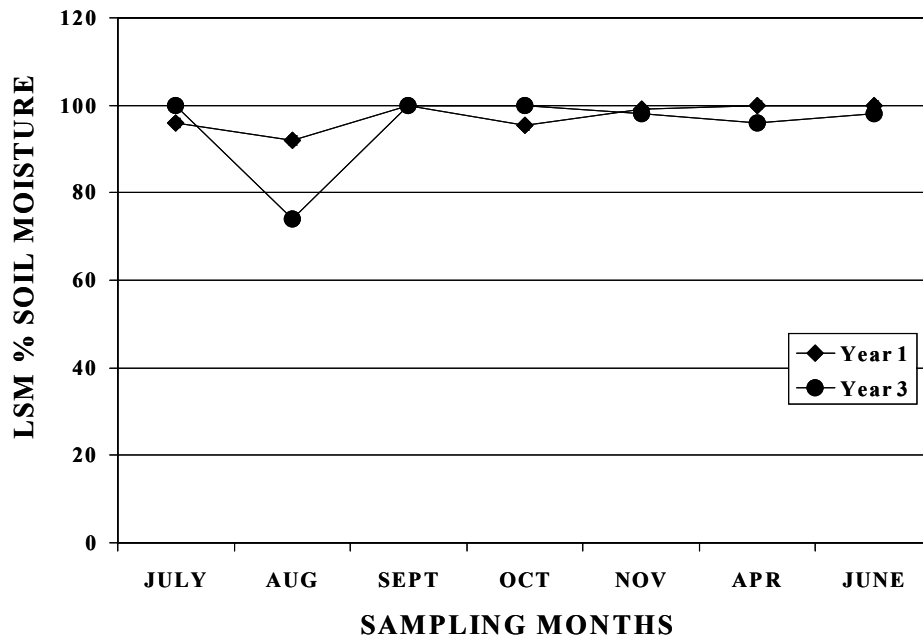


Figure 10: Least squares means percent soil moisture (i.e. saturation) for years 1 and 3.

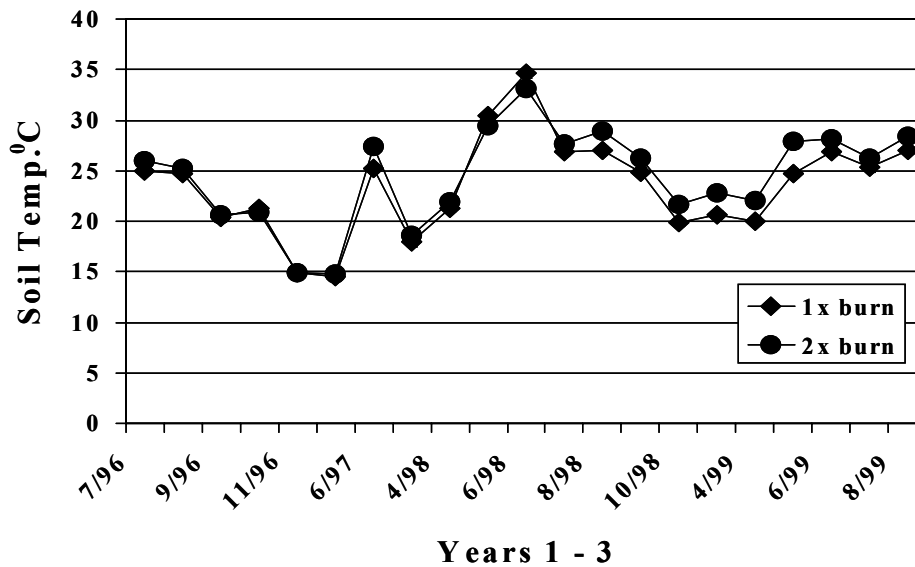


Figure 11: Least squares means for soil temperatures during years 1-3.

DISCUSSION

Species Richness

Ant species collected at Lake Ramsay were not unique to the sites. Researchers have reported these species from similar habitats in other portions of the longleaf pine ecosystem (Carter 1962, Deyrup et al. 2000, Folkerts 1993, Lubertazzi 1999). Twenty-three of 26 species collected in this study were reported in Lubertazzi's (1999) study using pitfall trap sampling in a longleaf pine flatwoods in the Apalachicola National Forest. The fauna was split in my study between eastern and southeastern species each contributing 38.5% and a little less than a quarter introduced from the tropics. None are savanna dependent or restricted, but I considered four species as representatives of this habitat because they are commonly found in coastal pine savannas: *Pheidole dentata*, *Pheidole dentigula*, *Pheidole metallescens*, and *Trachymyrmex septentrionalis*. *Crematogaster ashmeadi*, though not savanna dependent, nests in branches of longleaf pine and is regarded as a primary food of the endangered red-cockaded woodpecker (Hess and James 1998, Tschinkel and Hess 1999). My *Crematogaster ashmeadi* specimens may be a new species closely related to *Crematogaster ashmeadi* (Deyrup unpublished). This cryptic species is dependent on fire-maintained longleaf pine habitats. Additionally, there were five species common to coastal plain habitats which include longleaf pine savannas: *Camponotus castanea*, *Crematogaster pilosa*, *Monomorium viride*, *Polygerus lucidus*, and *Strumigenys louisianae*. Two species, *Aphaenogaster carolinensis* and *Paratrechina faisonensis* are not usually found in open, grassy locations. In fact, *Aphaenogaster carolinensis* was only collected prior to the first applied fire when

leaf litter was dense. An annotated checklist of ant species collected at Lake Ramsay is given in the Appendix 2.

Although longleaf pine savannas have been estimated to contain 4,000-5,000 species of insects (Folkerts 1993), they do not appear to contain a diverse fauna of ants. If the 26 species collected in this study are representative of ant diversity in longleaf pine savannas then ants comprise only half of a percent of the total insect diversity estimated by Folkerts. Such a small number is difficult to accept considering ants are one of the most diverse groups of insects in terrestrial ecosystems (Whitford et al. 1999). However, if nearctic and neotropical diversity are compared, North American ant species represent only 6.5% of the world's known ant diversity while the neotropics possess 24.5% (Hölldobler and Wilson 1990). The species richness I reported was similar to another study in a longleaf pine flatwoods savanna in the Apalachicola National Forest where between 26 and 32 species of ants were collected with pitfall traps (Lubertazzi 1999). An earlier survey conducted in Florida longleaf pine flatwoods reported 29 native ant species prior to *S. invicta* invasion; wetter habitats such as bayheads and marshes had 25 and 11 species respectively (Van Pelt 1956). Possibly, species richness observed in these habitats was limited to some extent by collection method. However, pitfall traps do provide a good estimate of ant species richness and relative abundance (Luff 1975, Wang et al. 2001). The fact that only 29 species were collected prior to *S. invicta* invasion suggests the ant fauna was not historically rich, nor was it dramatically affected or depressed by *S. invicta*.

Is the habitat somehow restrictive to many potential species? Culver (1974) attributed lower ant species richness in grasslands to having less structural diversity than

forests. Structurally simple habitats like grasslands foster territorial aggression. Aggression was prohibitive to a diverse ant community. Structural diversity of the plant community in forests limited competition among ants by providing greater foraging niches (Culver 1974). Lubertazzi (1999) also found a negative correlation between ant species richness and total herbaceous ground cover. In the above study where between 26 and 32 species were collected from a flatwoods savanna, species richness increased to 47 as the percentage of woody plant cover increased on research plots. However, the number of species found at Lake Ramsay was similar to the number of species found in an Appalachian oak/pine forest. Thirty-one species of ants were collected using pitfall traps (Wang et al. 2001). Researchers (Wang et al. 2001) reported a negative correlation between soil moisture and ant species richness. Species collected from wetter sites had lower relative abundances. A desert grassland study looking at ant species richness as an indicator of disturbance stress, concluded soil qualities influenced species richness more than disturbance and vegetative composition (Whitford et al. 1999). Twenty-one species of ants were collected in pitfall traps. Ant species richness was more similar on chronically disturbed sites and adjacent undisturbed sites than disturbed sites in other locations. All adjacent sites shared soils of the same soil series. There is a positive correlation between structural diversity of vegetation and ant species richness (Culver 1974), but many factors including latitude, soil type, and moisture influence the complexity of a habitat such that a desert grassland may contain as many species as a eastern, deciduous forest or a longleaf pine savanna.

Peanut butter and honey baited vials were used to monitor recolonization of *S. invicta* in plots treated with Amdro[®], but this collecting method added three species not

sampled by pitfall traps: *Tapinoma sessile*, *Crematogaster missouriensis*, and *Pseudomyrmex pallida*. Red imported fire ants outnumbered other species 66:1 in abundance at these vials. *Solenopsis invicta* are efficient foragers, and workers recruit to food sources in large numbers (Vinson 1994). No other exotics were collected at baits until after the first applied fire. Prior to fire application, *Crematogaster pilosa*, a native, mainly arboreal species, was collected in relatively large numbers (e.g. 34 *C. pilosa* individuals to 3200 *S. invicta* individuals). Once treatments were applied two other exotics, *Pheidole flavens* and *Brachymyrmex musculus* were frequently found at baits and *Paratrechina faisonensis*, a native, was always present in low numbers. *Paratrechina faisonensis* was routinely collected along with *S. invicta* at baits. *Solenopsis invicta* preferred peanut butter baits 85%-95% to honey baits. Recruitment to honey increased during August and September.

Some interesting trends in numbers of ants collected with baited vials during the three-year study period are shown in Table 11. For example, numbers of ants collected in treatment 2xNA when compared to 1xNA reveal the effect of repeated fires. That is, applying fire in two consecutive years led to a noticeable reduction in native species' relative abundances and a dramatic increase in *S. invicta* and other exotics. In contrast, a single applied fire in two years produced the greatest relative increase in natives regardless of exotics. As mentioned previously, there is a tendency for species richness to decrease as habitat complexity decreases. Repeated removal of standing vegetation and leaf litter not only simplified my savanna sites, but altered abiotic components as well. There is an immediate, observable impact on certain species due to simplification of the habitat by disturbance. Environmental parameters measured in my study were not

associated with ant species differences due to treatments, which leads me to believe that any influence on those species from changes in abiotic features is a slower process.

Table 11: Number of ants collected in peanut butter vials during years 1-3.

Treatment		Year		
Ant type		1	2	3
1xA	Native	7	62	117
	Exotic	0	1	9
	<i>S. invicta</i>	11,376	5,330	30,300
1xNA	Native	29	40	442
	Exotic	0	160	144
	<i>S. invicta</i>	12,863	32,434	41,058
2xA	Native	1	58	151
	Exotic	0	190	165
	<i>S. invicta</i>	7,586	7,058	26,284
2xNA	Native	142	223	31
	Exotic	0	9	420
	<i>S. invicta</i>	13,301	30,631	55,845

Beetles

Species richness of carabid and identified staphylinid beetles was comparable to my ant species richness. Even though pitfall traps are preferred for collection of ground beetles, my data showed 10 of 28 species collected were represented by a single individual. Other researchers have reported low numbers of individuals collected from grasslands and often present data on the most abundant species or statistics calculated at the family level (Collett and Neumann 1995, Hu and Frank 1996, Morris and Rispin 1987, Rickard 1970, Rushton et al. 1990). Staphylinid beetles are evidently much more abundant or readily collected (Collett and Neumann 1995, Hu and Frank 1996, Morris and Rispin 1987). Baars (1979) looked at different constructions and arrangements of pitfall traps to evaluate catches in relation to mean densities of carabids. He found shape

and number of traps did not influence the amount of carabids caught, and concluded that an individual's activity level, not population density, influenced collection numbers.

Low numbers may also be attributed to establishment of *S. invicta* at Lake Ramsay. No pre-invasion surveys exist, so the validity of this possibility cannot be determined. Red imported fire ants are known to reduce populations of carabids and staphylinids (Hu and Frank 1996, Stoker et al. 1995). Invasion by *S. invicta* has led to simplification of invertebrate communities by replacement through competition and/or predation of predatory, phytophagous, and necrophagous species (Hu and Frank 1996, Stoker et al. 1995, Vinson 1994). Beetle species diversity from Amdro[®] treated pastures in Florida was compared to diversity on untreated pastures to reveal carabids and staphylinids were more abundant when *S. invicta* populations were reduced (Hu and Frank 1996). Low numbers of beetles collected at Lake Ramsay did not allow for analysis of treatment differences. Plots receiving a single applied fire and no Amdro[®] had a lower frequency of beetles collected than plots from the other three treatments. All beetles collected from that treatment were also collected from one or more of the other treatments, making it difficult to speculate about reasons for the discrepancy. Trophic and other life history information necessary for predicting impacts of management and restoration protocols is sparse for ground beetle species (Morris and Rispin 1987). However, I do not believe that fires applied to my research plots had a negative impact on beetle species because data from other studies have shown ground beetle species richness is not altered by burning fire maintained habitats (Hansen 1986, Rickard 1970).

Rushton et al (1989) studied ground beetle differences in grasslands under various stages of restoration and concluded soil moisture and ground cover (i.e. grass tussocks)

were the two factors that most influenced ground beetle communities. At Lake Ramsay, soil remained saturated except for parts of July and August at which time the upper layer was dry and cracked. I suspect in such a harsh environment most species resided in grass tussocks or along rises in the terrain where longleaf grow. Grass tussocks maintain a stable temperature and humidity more favorable to beetles than exposed soil (Curry 1994). Though Lake Ramsay savanna has a diverse plant community, grasses and forbs are not structurally diverse thereby limiting the number of available niches, especially in a litter reduced, fire-prone community.

Effects of Fire

Understanding the full effect of fire frequency on selected species is difficult because few places have been studied for long periods of time (Hermann et al. 1998). Variation in the literature about insect population responses to fire is not unusual. For example, in a study of fire effects on prairie arthropods one species of leafhopper increased in abundance after a prescribed fire, but another species in the same genus decreased (Harper et al. 2000). One possibility for these differences is organisms within a group exist at different strata within a habitat. For instance, if one organism forages mostly on the soil surface and another prefers to forage within litter, then the one closer to the soil will most likely be collected in pitfall traps prior to removal of litter by fire. This may explain before and after burn differences I encountered with my beetle data. Collections of ground beetles doubled following the first applied fire. Possibly, the rapid increase in relative abundance could reflect movement of individuals into the disturbed area or that removal of ground cover necessitated litter dwellers to forage on the soil surface in closer proximity to pitfall traps. However, after a fire both foragers occur

together and are collected at the soil surface. Supporting this idea, Hansen (1986) documented greater species diversity from pitfall samples on burned portions of a western rangeland compared to adjacent unburned areas for up to one month after the fire. The rapid increase in species richness and abundance is indicative of immigration from neighboring areas and exposure of resident species. “Increased knowledge of a wide range of organisms in the habitat will result in more effective application of prescribed fire and, consequently, better ecosystem management” (Hermann et al. 1998), but understanding species level responses to treatments requires knowledge of their life histories. Little information is known or published about grassland beetle species (Curry 1994), and for a majority of the species I collected, ant or beetle, it was difficult to find a key for identification or biological information.

Another explanation for disparity among research findings is that just as location differs among studies involving fire so do aspects of applied fires. Mortality and subsequent recolonization following fires depends on size and intensity of the fire and the extent to which the physical environment was altered (Willig and McGinley 1999). Nevertheless, predicting the impact of fire on inhabitants is difficult because, “no two fires are alike, and the outcome of fire depends on its timing, intensity, recurrence interval, and subsequent weather” (D’Antonio 2000). Historical frequencies of fires are regionally distinct, intensity of fires is not a constant, and differences in season of burn produce divergent results. Whether or not such details influence insect species composition is not fully realized. However, Lubertazzi’s (1999) survey of ant fauna in longleaf pine flatwoods revealed ant species richness is inversely related to plant species richness under different burning regimes. In his research, a fire frequency of three years

or less produced a diverse understory of grasses and forbs while a frequency of greater than three years produced a less diverse understory with more woody species. Ant diversity was lower in grassy areas and higher in wooded areas. Researchers have established that long intervals between fires encourage species that are varied in their response to fire. Periodic fires on the other hand generally do not produce major changes in populations (Hermann et al. 1998, Whelan et al. 1980).

Mortality of insects is influenced by time of year a fire is applied, time of day, speed and intensity of the fire, and patchiness of the burn. Mortality increases with greater time between fires because of an increase in generalist species and increased fuel loads which lead to a more intense fire. Land left unburned for many years will respond differently than land subjected to frequent fires (Hermann et al 1998). In areas with moderate fire frequencies and intensities, fire may have less of an effect than the environmental changes that occur as a result of fire (Anderson 1991, Anderson et al. 1989). These changes can directly impact individuals or act indirectly by influencing competitive interactions (Anderson 1991, Anderson et al. 1989). Environmental parameters in my study did not contribute to measured changes within ant species. Soil temperatures and leaf litter amounts were significantly different among treatments; however, neither environmental parameter contributed to differences detected in ant abundance among treatments. If this study had continued and soil temperatures among treatments continued to diverge such a difference might eventually impact resident species. Measurements on leaf litter accumulation validated the resiliency of a fire prone system to recover quickly. Plots receiving a single applied fire had 60% recovery of pretreatment litter dry weights two years later.

Because savannas have frequent fires, burns tend to be patchy (Platt 1994). Patchiness provides refugia for savanna organisms and soil temperatures are not hot enough to kill all inhabitants. Many soil and litter dwelling insects survive fire by escaping from the heat. Those populations often increase in size and number after fire due to increased food availability and disturbance to the area (Ahlgren 1974). For example, mound building ants rarely suffer reduction in numbers after fire. Disturbances that open areas are beneficial for mound building. This is particularly true for *S. invicta*, the only mound building species collected at Lake Ramsay. In Zimmer and Parmenter's (1998) study of harvester ant activity following fire, they found no change in number of colonies or number of foragers. They observed an increase in foraging territories and greater numbers of insect parts being brought into the colonies. In other *Pogonomyrmex* species they found the increase in soil temperature following fires led to an increase in surface activity by the ants.

A study focusing on changes in invertebrate soil fauna of longleaf pine habitats burned and unburned for 10 years found diversity was not different between treatments (Heyward and Tissot 1936). Approximately eight species of ants collected at Lake Ramsay are known to nest in soil: *Aphaenogaster carolinensis*, *Brachymyrmex depilis*, *Camponotus castanea*, *Hypoconerops opaciceps*, *Hypoconerops opacior*, *Monomorium viride*, *Pheidole dentata*, and *Trachymyrmex septentrionalis* (Appendix 2). Only *Hypoconerops opaciceps* and *Hypoconerops opacior* showed noticeable increases in numbers collected following treatments (Table 4). The other six species were, for the most part, collected at the same frequency throughout the study.

A review of the literature indicated another seven ant species to be litter dwellers, nesting in grass and woody debris (Appendix 2). Relative abundances of *Cyphomyrmex rimosus*, *Formica pallidefulva*, *Paratrechina faisonensis*, *Pheidole flavens*, and *Strumigenys louisianae* increased due to applied treatments (Table 4). The other two species *Pheidole dentigula* and *Ponera pennsylvanica* did not appear to be influenced by treatments. Andrew et al. (2000) designed a test to determine whether or not litter dwelling ant species found refuge in unburned portions of a habitat subjected to low-intensity experimental fires. Contrary to their initial hypothesis species richness and abundance of ants did not differ among refuges and exposed portions of the habitat. Additionally, predaceous species of carabids, coccinellids, and staphylinids were often in greater abundance in burned areas (Ahlgren 1974, Anderson et al. 1989, Van Amburg et al. 1981). These litter dwellers are not harmed by the loss of litter because they have the ability to burrow into soil, hide in grass tussocks, or fly short distances to escape fire.

Impact of Exotics

Abiotic elements between a species home range and introduced range are often similar. In South America *S. invicta* is found in disturbed cerrado. Cerrado is similar to longleaf pine savannas in that it has fire-adapted vegetation, acidic soils, and lateritic hardpans (Wojcik 1986). Lake Ramsay does not have lateritic hardpans, but the fragipan layer does keep water from filtering through the soil. Invasive exotics are often superior competitors making abiotic factors, not native species, the limiting factor in successful invasions (Culver 1974, Holway 1998). Temperate ant species do not mobilize as quickly or in as great numbers as tropical species (Culver 1974). Thus, fire frequencies may be a more important regulator of *S. invicta* populations than native species. I found both *S.*

invicta and other exotic ants responded positively to disturbance by fire. Exotics from 2x fire plots continued to increase in year 3 while those species from 1x fire plots began to decrease in abundance.

Our understanding of disturbance as it relates to invasion is not complete. This is a complex problem involving distributions, abundances, and interaction of well-known and not so well known species (Orians 1989). Disturbances can make a habitat vulnerable to invasion by exotics by increasing available resources and potentially decreasing the number of competing species (Hobbs 1989, Holway 1998). Camilo and Philips (1990) reported a negative correlation between native ant species and *S. invicta* in disturbed and undisturbed field plots. Pitfall traps on plots without *S. invicta* or disturbance had 21 native ant species while those plots with disturbance only had 14 species and those with *S. invicta* and disturbance collected only five native ant species (Camilo and Philips 1990). Disturbance by fire favored exotic species in my research. Prior to treatments *S. invicta* and two native species, *Paratrechina faisonensis* and *Crematogaster pilosa* were the three dominant species. After treatments, the two dominant native species were displaced by exotics, *Cyphomyrmex rimosus* and *Pheidole flavens*. Initial application of treatments did influence the order in which species were distributed. Among the four treatment combinations all species responded the same with respect to rank order of abundance. Species composition became more homogenous among treatments once treatments were applied.

Exotic ant species represented approximately one-fourth of species collected, but comprised 98% of the individuals collected in pitfall traps and 98.5% of individuals sampled by baits. Relative abundance of exotics increased due to managed late growing

season fires. Between years 2 and 3, exotic species' abundance (excluding *S. invicta*) decreased on 1x fire plots and increased on 2x fire plots. Exotic ant species' abundances were reduced in treatments receiving Amdro[®]. Native species did not experience a significant change in abundance due to treatments. My results imply that land managers can at least maintain existing native species and minimize exotics by burning less often than annually. Amdro[®] does not appear to be necessary for maintenance of native species within a disturbance regime.

A descriptive examination of my data revealed some interesting trends. First treatments with two fires and no Amdro[®] had the greatest percent increase in exotic species abundance and an equivalent decrease in native species abundance. Therefore, if land managers burned annually one would expect exotics might eventually replace natives. Secondly, treatments with one fire and no Amdro[®] had the least number of natives collected, but natives experienced an 8-fold increase in abundance from pretreatment counts. Also, this treatment had the same ratio of exotic to native species as treatments with two fires and Amdro[®]. This could be coincidental, but applying fire once without chemical reduction of exotics produced the same low ratio of exotics to natives as applying fire twice with chemicals applied.

Past efforts to eradicate red imported fire ants have probably done greater harm to native species than to *S. invicta* (Bright 1999, Davidson and Stone 1989). When Mirex, a broad-spectrum insecticide, was broadcast throughout the Southeast in the late 50s and early 60s it not only decreased *S. invicta* populations, but also decimated many native competitors of *S. invicta* (Davidson and Stone 1989). Not surprisingly, one year following initial applications of Mirex a 10-fold increase in *S. invicta* abundance was

observed. I found Amdro[®] treated plots with two applied fires had 2.5 times more *S. invicta* collected after treatment applications than Amdro[®] treated plots with a single applied fire. Plots not receiving Amdro[®] only had a 1.4-fold difference between fire frequencies. The greater proportional increase in Amdro[®] treated plots compared to non-Amdro[®] plots was reminiscent of the situation with Mirex where suppression was followed by an even greater resurgence of *S. invicta* than previously experienced. In addition, applying fire both years resulted in significantly more *S. invicta* collected than applying fire once. The combination of suppression (Amdro[®]) with repeated disturbance (two applied fires) amplified *S. invicta* abundance. The observed difference would be worth pursuing in a study designed to determine whether or not *S. invicta* suppression leads to greater proportional increases in abundance with each applied disturbance.

Management Implications

The developing theme from the data suggests that if Lake Ramsay longleaf pine savanna is managed appropriately, existing native species can maintain their presence in the habitat along with the invasive exotics. Attempts to suppress or eradicate exotic ant species would probably not be successful and could actually increase their abundance in savannas. The fact that applying late growing season fires two consecutive years with Amdro[®] applications was not different with respect to relative abundance of *S. invicta* or native ant species compared to applying a single late growing season fire and not applying Amdro[®] has significant management implications. Use of chemicals to control unwanted species is costly and maybe detrimental to some nontarget species especially if an appropriate fire frequency can accomplish the same result. According to Deyrup et al. (2000), "The greatest impact [of exotics] on native species must be in semi-disturbed

areas...areas that may be important for certain native species even though the original ecosystem has been strongly modified”. Restoring and maintaining endangered ecosystems from a historical perspective is important (Hermann et al. 1998), but viewing the situation in light of present day species composition that include exotic species is equally important. Biological invasions have become part of “normal community processes” as a result of human activity so understanding the impact of introductions and extinctions on an ecosystem is crucial to “scientifically sound and effective” management (Ernest and Brown 2001, Hobbs 1989).

REFERENCES

- Lake Ramsay Savannah. **1992**. A Management Plan for Lake Ramsay Savannah Wildlife Management Area.
- Ahlgren, I. F. 1974**. The effect of fire on soil organisms. *In* Fire and Ecosystems, pp. 47-71. Edited by T. T. Kozlowski: Academic Press.
- Ali, A. D., T. E. Reagan, and J. L. Flynn. 1984**. Influence of selected weedy and weed-free sugarcane habitats on diet composition and foraging activity of the imported fire ant (Hymenoptera: Formicidae). *Environmental Entomology* 13: 1037-1041.
- Amacher, M. C., W. J. Day, B. A. Schumacher, P. M. Walthall, and B. J. Miller. 1989**. A Guide to the Classification of Soils of Louisiana. Louisiana Agriculture Experiment Station Bulletin 803.
- Anderson, A. N. 1990**. The use of ant communities to evaluate change in Australian terrestrial ecosystems: A review and a recipe. *Proceedings of the Ecological Society of Australia* 16: 347-357.
- Anderson, A. N. 1991**. Responses of ground-foraging ant communities to three experimental fire regimes in a savanna forest of tropical Australia. *Biotropica* 23: 575-585.
- Anderson, R. C., T. Leahy, and S. S. Dhillion. 1989**. Numbers and biomass of selected insect groups on burned and unburned sand prairie. *The American Midland Naturalist* 122: 151-162.
- Andrew, N., L. Rodgerson, and A. York. 2000**. Frequent fuel-reduction burning: The role of logs and associated leaf litter in the conservation of ant biodiversity. *Austral-Ecology* 25: 99-107.
- Apperson, C. S., R. B. Leidy, and E. E. Powell. 1984**. Effects of Amdro on the red imported fire ant (Hymenoptera: Formicidae) and some nontarget ant species and persistence of Amdro on a pasture habitat in North Carolina. *Journal of Economic Entomology* 77: 1012-1018.
- Baars, M. A. 1979**. Catches in pitfall traps in relation to mean densities of carabid beetles. *Oecologia* 63: 25-46.
- Beattie, A. J. 1989**. The effects of ants on grasslands. *In* Grassland Structure and Function: California Annual Grasslands, pp. 105-116. Edited by L. F. Huenneke and H. Mooney. Netherlands: Kluwer Academic Publishers.
- Bendell, J. F. 1974**. Effects of fire on birds and mammals. *In* Fire and Ecosystems, pp. 73-138. Edited by T. T. Kozlowski: Academic Press.

- Bright, C. 1999.** Crawling out of the pipe: the hazardous waste that makes more of itself. *World Watch* 12: 22-33.
- Brown, W. L. 1953.** Revisionary studies in the ant tribe Dacetini. *The American Midland Naturalist* 50: 1-137.
- Buren, W. F., G. E. Allen, W. H. Whitcomb, F. E. Lennartz, and R. N. Williams. 1974.** Zoogeography of the imported fire ants. *Journal of the New York Entomological Society* 82: 113-124.
- Camilo, G. R. and S. A. Philips. 1990.** Evolution of ant communities in response to invasion by the fire ant *Solenopsis invicta*. *In Applied Myrmecology. A world Perspective*, pp. 190-198. Edited by R. K. Vander Meer, K. Jaffe and A. Cedeno. Boulder: Westview Press.
- Cancelado, R. and T. R. Yonke. 1970.** Effect of prairie burning on insect populations. *Journal of the Kansas Entomological Society* 43: 274-281.
- Canter, L. W. 1981.** Final programmatic environmental impact statement. USDA APHIS-ADM Report 81-01-F.
- Carroll, J. F. 1975.** Biology and Ecology of Ants of the Genus *Aphaenogaster* in Florida. Dissertation. University of Florida, Gainesville.
- Carter, W. G. 1962.** Ant distribution in North Carolina. *The Journal of the Mitchell Society* 78: 150-204.
- Chapman, H. H. 1932.** Is the longleaf type a climax? *Ecology* 13: 328-334.
- Christensen, N. L. 1978.** Fire regimes in southeastern ecosystems. *In Fire Regimes and Ecosystem Properties*, pp. 112-136. USDA General Technical Report WO-26.
- Colinvaux, P. A., K.B. Liu, P. DeOliveira, M.B. Bush, M.C. Miller, and M.S. Kannan. 1996.** Temperature depression in the lowland tropics in glacial times. *Climatic Change* 32: 19-33.
- Collett, N. G. and F. G. Naumann. 1995.** Effects of two spring prescribed fires on epigeal Coleoptera in dry sclerophyll eucalypt forest in Victoria, Australia. *Forest Ecology and Management* 76: 69-85.
- Creighton, W. S. 1950.** The ants of North America. *Bulletin of the Museum of Comparative Zoology* 104: 1-585.
- Culver, D. C. 1974.** Species packing in Caribbean and north temperate ant communities. *Ecology* 55: 974-988.

- Curry, J. P. 1994.** Grassland Invertebrates: Ecology, Influence on Soil Fertility and Effects on Plant Growth. Chapman and Hall, New York.
- D'Antonio, C. M. 2000.** Fire, plant invasions, and global changes. *In* Invasive Species in a Changing World, pp. 65-93. Edited by H. A. Mooney and R. J. Hobbs. Washington, D.C.: Island Press.
- Davidson, N. A. and N. D. Stone. 1989.** Imported fire ants. *In* Eradication of Exotic Pests, pp. 196-218. Edited by D. L. Dahlsten and R. Garcia. New Haven: Yale University Press.
- Deyrup, M.** Formal recognition of a new species of *Crematogaster* from southeastern pinelands (Hymenoptera: Formicidae). Unpublished.
- Deyrup, M., L. Davis, and S. Cover. 2000.** Exotic ants in Florida. Transactions of the American Entomological Society 126: 293-326.
- Deyrup, M. A., N. Carlin, J. Trager, and G. Umphrey. 1988.** A review of the ants of the Florida Keys. Florida Entomologist 71: 163-176.
- Dobson, A. and B. Czech. 1997.** Distribution and causation of species endangerment in the United States. Science 277: 1116-1117.
- DuBois, M. B. 1986.** A revision of the native new world species of the ant genus *Monomorium* (*minum* group) (Hymenoptera: Formicidae). The University of Kansas Science Bulletin 53: 65-119.
- Ernest, S. K. M. and J. H. Brown. 2001.** Delayed compensation for missing keystone species by colonization. Science 292: 101-104.
- Everett, R. A. 2000.** Patterns and pathways of biological invasions. Trends in Ecology and Evolution 15: 177-178.
- Folkerts, G. W., M. A. Deyrup, and D. C. Sisson. 1993.** Arthropods associated with xeric longleaf pine habitats in the southeastern United States: A brief overview. Proceedings of the Tall Timbers Fire Ecology Conference 18: 159-192.
- Frost, C. C. 1993.** Four centuries of changing landscape patterns in the longleaf pine ecosystem. Proceedings of the Tall Timbers Fire Ecology Conference 18:17-43.
- Hahn, D. A. and W. R. Tschinkel. 1997.** Settlement and distribution of colony-founding queens of the arboreal ant, *Crematogaster ashmeadi*, in a longleaf pine forest. Insectes sociaux 44: 323-336.

- Hansen, J. D. 1986.** Comparison of insects from burned and unburned areas after a range fire. *Great Basin Naturalist* 46: 721-727.
- Harper, M. G., C. H. Dietrich, R. L. Larimore, and P. A. Tessene. 2000.** Effects of prescribed fire on prairie arthropods: An enclosure study. *Natural Areas Journal* 20:325-335.
- Hermann, S. M., T. Van Hook, R. W. Flowers, L. A. Brennan, J. S. Glitzenstein, D. R. Streng, J. L. Walker, and R. L. Myers. 1998.** Fire and biodiversity: Studies of vegetation and arthropods. *Transactions of the North American Wildlife and Natural Resources Conference* 63: 384-401.
- Hess, C. A. and F. C. James. 1998.** The diet of the red-cockaded woodpecker in the Apalachicola National Forest. *Journal of Wildlife Management* 62: 509-517.
- Heyward, F. 1939.** The relation of fire to stand composition of longleaf pine forests. *Ecology* 20: 287-304.
- Heyward, F. and A. N. Tissot. 1936.** Some changes in the soil fauna associated with forest fires in the longleaf pine region. *Ecology* 17: 659-666.
- Hobbs, R. J. 1989.** The nature and effects of disturbance relative to invasions. *In* *Biological Invasions: A Global Perspective*, pp. 389-405. Edited by J. A. Drake, H. A. Mooney, F. di Castri, R. H. Groves, F. J. Kruger, M. Rejmanek and M. Williamson. New York: John Wiley & Sons, Ltd.
- Holldobler, B. and E. O. Wilson. 1990.** *The Ants*. Belknap Press, Cambridge, MA.
- Holway, D. A. 1998.** Factors governing rate of invasion: a natural experiment using Argentine ants. *Oecologia* 115: 206-212.
- Howard, F. W. and A. D. Oliver. 1978.** Arthropod populations in permanent pastures treated and untreated with Mirex for red imported fire ant control. *Environmental Entomology* 7: 901-903.
- Hu, G. Y. and J. H. Frank. 1996.** Effect of the red imported fire ant (Hymenoptera: Formicidae) on dung-inhabiting arthropods in Florida. *Environmental Entomology* 25: 1290-1296.
- Hulbert, S.H. 1984.** Pseudoreplication and the design of ecological field experiments. *Ecological Monograph* 54: 187-211.
- Johnson, C. 1988.** Species identification in the eastern *Crematogaster* (Hymenoptera: Formicidae). *Journal of Entomological Science* 23: 314-332.

- Johnson, N. E. and R. S. Cameron. 1969.** Phytophagous ground beetles. *Annals of the Entomological Society of America*- 62: 909-913.
- Kalisz, P. J. and E. L. Stone. 1984.** The longleaf pine islands of the Ocala National Forest, Florida: A soil study. *Ecology* 65: 1743-1754.
- Lavigne, R.J. and M.K. Campion. 1978.** The effect of ecosystem stress on the abundance and biomass of Carabidae (Coleoptera) on the shortgrass prairie. *Environmental Entomology* 7: 88-92.
- Littell, R. C., G. A. Milliken, W. W. Stroup, and R. D. Wolfinger. 1996.** SAS System for Mixed Models. SAS Institute, Inc., Cary, NC.
- Lovell, D. L., R. A. Henderson, and E. A. Howell. 1982.** The response of forb species to seasonal timing of prescribed burns in remnant Wisconsin prairies. *In North American Prairie Conference*, pp. 11-15. Edited by R. Brewer. Michigan University.
- Lubertazzi, D. 1999.** Ant (Formicidae) Community Change Across a Vegetational Gradient in North Florida Longleaf Pine (*Pinus palustris*) Flatwoods. Dissertation. Biological Sciences Department, Florida State University.
- Luff, M. L. 1975.** Some features influencing the efficiency of pitfall traps. *Oecologia* 19: 345-357.
- Lynch, J. F. and A. K. Johnson. 1988.** Spatial and temporal variation in the abundance and diversity of ants (Hymenoptera: Formicidae) in the soil and litter layers of a Maryland forest. *The American Midland Naturalist* 119: 31-44.
- MacKay, W. P. and R. S. Anderson. 1991.** New distributional records for the ant genus *Ponera* (Hymenoptera: Formicidae) in North America. *Journal of the New York Entomological Society* 99: 696-699.
- Magurran, A. E. 1988.** *Ecological Diversity and Its Measurement*. Princeton University Press, Princeton.
- Maina, G. G. and H. F. Howe. 2000.** Inherent rarity in community restoration. *Conservation Biology* 14: 1335-1340.
- McCune, B. and M. J. Mefford. 1999.** PC-ORD. Multivariate Analysis of Ecological Data. MjM Software Design, Geneden Beach, OR.
- Milliken, G. A. and D. E. Johnson. 1989.** *Analysis of Messy Data: Nonreplicated Experiments*. Chapman & Hall, New York.

- Morrill, W. L. 1992.** Gound beetles (Coleoptera: Carabidae) in Georgia grasslands. *Journal of Agricultural Entomology* 9: 179-188.
- Morris, M. G. and W. E. Rispin. 1987.** Abundance and diversity of the coleopterous fauna from a calcareous grassland under different cutting regimes. *Journal of Applied Ecology* 24: 451-465.
- Naves, M. A. 1985.** A monograph of the genus *Pheidole* in Florida (Hymenoptera: Formicidae). *Insecta Mundi* 1: 53-97.
- Noss, R. 1988.** The longleaf pine landscape of the Southeast: Almost gone and almost forgotten. *Endangered Species UPDATE* 5: 1-8.
- Noss, R. F. 1995.** What should endangered ecosystems mean to the wildlands project? *Wild Earth Winter*: 20-28.
- Noss, R. F., E. T. I. La Roe, and J. M. Scott. 1995.** Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. National Biological Services Report 28.
- Oosterheld, M., J. Loreti, M. Semmartin, and J. M. Paruelo. 1999.** Grazing, fire, and climate effects on primary productivity of grasslands and savannas. *In Ecosystems of Disturbed Ground*, pp. 287-306. Edited by L. R. Walker. New York: Elsevier.
- Olson, M.S. and W.J. Platt. 1995.** Effects of habitat and growing season fires on resprouting of shrubs in longleaf pine savannas. *Vegetatio* 119: 101-118.
- Orians, G. H. 1989.** Site characteristics favoring invasions. *In Biological Invasions*, pp. 133-148. Edited by J. A. Drake, H. A. Mooney, F. di Castri, R. H. Groves, F. J. Kruger, M. Rejmanek and M. Williamson. New York: John Wiley & Sons.
- Outcalt, K. W. and R. M. Sheffield. 1996.** The Longleaf Pine Forest: Trends and Current Conditions. Southern Research Station, Asheville, NC.
- Panzer, R. 1988.** Managing prairie remnants for insect conservation. *Natural Areas Journal* 8: 83-90.
- Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000.** Environmental and economic costs of nonindigenous species in the United States. *BioScience* 50: 53-65.
- Platt, W. J. 1994.** Evolutionary Models of Plant Population/Community Dynamics and Conservation of Southeastern Pine Savannas. *In North American Conference on Barrens and Savannas*, pp. 265-273. Edited by J.S. Fralish, R.C. Anderson, J.E. Ebinger, and R. Szafoni. Normal, IL: Illinois State University, The Nature

Conservancy, United States Environmental Protection Agency, Illinois Department for Conservation, and Society for Ecological Restoration.

- Platt, W. J. 1999.** Southeastern pine savannas. *In* Southeastern Pine Savannas, pp. 23-51. Edited by R. C. Anderson, J. S. Fralish and J. Baskin. Cambridge, England: Cambridge University Press.
- Porter, S. D. and D. A. Savignano. 1990.** Invasion of polygyne fire ants decimates native ants and disrupts arthropod community. *Ecology* 71: 2095-2106.
- Rickard, W. H. 1970.** Ground dwelling beetles in burned and unburned vegetation. *Journal of Rangeland Management* 23: 293-297.
- Ricklefs, R. W. 1990.** *Ecology*. W.H. Freeman and Company, New York.
- Rushton, S. P., M. D. Eyre, and M. L. Luff. 1990.** The effects of scrub management on the ground beetles of oolitic limestone grassland at Castor Hanglands National Nature Reserve, Cambridgeshire, UK. *Biological Conservation* 51: 97-111.
- Rushton, S. P., M. L. Luff, and M. D. Eyre. 1989.** Effects of pasture improvement and management on the ground beetle and spider communities of upland grasslands. *Journal of Applied Ecology* 26: 489-503.
- SAS Institute. 2001.** SAS System for Microsoft Windows (TS2MO). SAS Institute, Cary, NC.
- Simmons, T., R. Myers, P. Seamon, and J. Selby. 1995.** Bugs in your burn? *Rx Fire Notes* 4: 1-5.
- Snelling, R. R. and J. T. Longino. 1992.** Revisionary notes on the fungus-growing ants of the genus *Cyphomyrmex rimosus* group (Hymenoptera: Formicidae: Attini). *In* *Insects of Panama and MesoAmerica*, pp. 479-494. Edited by D. Quintero and A. Aiello. Oxford: Oxford University Press.
- Springett, J. A. 1976.** The effect of prescribed burning on the soil fauna and on litter decomposition in Western Australian forests. *Australian Journal of Ecology* 1: 77-82.
- Stoker, R. L., W. E. Grant, and S. B. Vinson. 1995.** *Solenopsis invicta* (Hymenoptera: Formicidae) effect on invertebrate decomposers of Carrion in Central Texas. *Environmental Entomology* 24: 817-822.
- Streng, D.R., J.S. Glitzenstein, and W.J. Platt. 1993.** Evaluating season of burn in longleaf pine forests: a critical literature review and some results from an ongoing long-term study. *Proceedings Tall Timbers Fire Ecology Conference* 18: 227-263.

- Taylor, R. W. 1967.** A monographic revision of the ant genus *Ponera* Latreille (Hymenoptera: Formicidae). Pacific Insects Monograph 13: 1-114.
- Thompson, C. R. 1989.** The thief ants, *Solenopsis molesta* group, of Florida (Hymenoptera: Formicidae). Florida Entomologist 72: 268-283.
- Trager, J. C. 1984.** A revision of the genus *Paratrechina* (Hymenoptera: Formicidae) of the continental United States. Sociobiology 9: 1-162.
- Tschinkel, W. R. and C. A. Hess. 1999.** Arboreal ant community of a pine forest in northern Florida. Annals of the Entomological Society of America 92: 63-70.
- Van Amburg, G. L., J. A. Swaby, and R. H. Pemble. 1981.** Response of arthropods to a spring burn of a tallgrass prairie in northwestern Minnesota. In The 6th North American Prairie Conference, pp. 240-243. Edited by R.L. Stuckey and K.J. Reese. Columbus, OH: College of Biological Sciences, Ohio State University.
- Van Mantgem, P., M. Schwartz, and M. B. Keifer. 2001.** Monitoring fire effects for managed burns and wildfires: Coming to terms with pseudoreplication. Natural Areas Journal 21: 266-273.
- Van Pelt, A. F. J. 1956.** The ecology of the ants of the Welaka Reserve, Florida (Hymenoptera: Formicidae). The American Midland Naturalist 56: 358-387.
- Vinson, S. B. 1994.** Impact of the invasion of *Solenopsis invicta* (Buren) on native food webs. In Exotic Ants, pp. 240-258. Edited by D. F. Williams. Boulder: Westview Press.
- Vinson, S. B. and L. Greenberg. 1986.** The biology, physiology, and ecology of imported fire ants. In Economic Impact and Control of Social Insects, pp. 193-226. Edited by S. B. Vinson. New York: Praeger.
- Vinson, S. B. and A. A. Sorensen. 1986.** Imported fire ants: Life history and impact. Texas Department of Agriculture, Austin, TX.
- Wahlenberg, W. G. 1946.** Longleaf Pine. Charles Lathrap Pack Forestry Foundation, Washington, D.C.
- Wang, C., J. Strazanac, and L. Butler. 2001.** A comparison of pitfall traps with bait traps for studying leaf litter ant communities. Journal of Economic Entomology 94: 761-765.
- Wang, C., J. S. Strazanac, and L. Butler. 2001.** Association between ants (Hymenoptera: Formicidae) and habitat characteristics in oak-dominated mixed forests. Environmental Entomology 30: 842-848.

- Whelan, R. J., W. Langedyk, and A. S. Pashby. 1980.** The effects of wildfire on arthropod populations in Jarrah-*Banksia* woodland. *Western Australia Naturalist* 14: 214-220.
- Whitcomb, W. H., H. A. Denmark, A. P. Bhatkar, and G. L. Greene. 1972.** Preliminary studies of the ants of Florida soybean fields. *Florida Entomologist* 55: 129-142.
- Whitford, W. G., J. Van Zee, M. S. Nash, W. E. Smith, and J. E. Herrick. 1999.** Ants as indicators of exposure to environmental stressors in North American desert grasslands. *Environmental Monitoring and Assessment* 54: 143-171.
- Wilcove, D. S. and L. Y. Chen. 1998.** Management costs for endangered species. *Conservation Biology* 12: 1405-1407.
- Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998.** Quantifying threats to imperiled species in the United States. *BioScience* 48: 607-615.
- Willig, M. R. and M. A. McGinley. 1999.** The response of animals to disturbance and their roles in patch generation. *In Ecosystems of Disturbed Ground*, pp. 633-657. Edited by L. R. Walker. New York: Elsevier.
- Wilson, N. L. and A. D. Oliver. 1969.** Food habits of the imported fire ants in pastures and pine forest areas in southeastern Louisiana. *Journal of Economic Entomology* 62: 1268-1271.
- Wojcik, D. P. 1986.** Observations on the biology and ecology of fire ants in Brazil. *In Fire Ants and Leaf-Cutting Ants*, pp. 88-103. Edited by C. S. Lofgren and R. K. Vander Meer. Boulder: Westview Press.
- Wojcik, D. P., C. R. Allen, R. J. Brenner, E. A. Forays, D. P. Jouvenaz, and R. S. Lutz. 2001.** Red imported fire ants: Impact on biodiversity. *American Entomologist* 47: 16-23.
- Yensen, N., E. Yensen, and D. Yensen. 1980.** Intertidal ants from the Gulf of California, Mexico. *Annals of the Entomological Society of America* 73: 266-269.
- Zedler, J. B. 2000.** Progress in wetland restoration ecology. *Trends in Ecology and Evolution* 15: 402-407.
- Zimmer, K. and R. R. Parmenter. 1998.** Harvester ants and fire in a desert grassland: ecological responses of *Pogonomyrmex rugosus* (Hymenoptera: Formicidae) to experimental wildfires in central New Mexico. *Environmental Entomology* 27: 282-287.

**APPENDIX 1. FEDERALLY LISTED AND CANDIDATE SPECIES
ASSOCIATED WITH THE LONGLEAF PINE ECOSYSTEM (NOSS ET AL.
1995)**

Listed Species - Plants

Apalachicola rosemary (*Conradina glabra*)
Pigeon-wing (*Clitoria fragrans*)
Beautiful pawpaw (*Deeringothamnus pulchellus*)
Rugel's pawpaw (*Deeringothamnus rugellii*)
Scrub mint (*Dicerandra frutescens*) .
Scrub buckwheat (*Eriogonum longifolium* var. *gnaphalifolium*)
Harper's beauty (*Harperocallis flava*)
Rough-leaf loosestrife (*Lysimachia asperulifolia*)
Britton's bear-grass (*Nolina brittonia*)
Godfrey's butterwort (*Pinguicula ionantha*)
Chapman's rhododendron (*Rhododendron chapmanii*)
Michaux's sumac (*Rhus michauxii*)
Green pitcherplant (*Sarracenia oreophila*)
Chaffseed (*Schwalbea americana*)
Gentian pinkroot (*Spigelia gentianoides*)
Cooley's meadowrue (*Thalictrum cooleyi*)
Clasping warea (*Warea amplexifolia*)
Carter's warea (*Warea carteri*)

Listed Species - Reptiles

Gopher tortoise (*Gopherus polyphemus*)
Sand skink (*Neoceps reynoldsi*)
Indigo snake (*Drymarchon corais couperi*)
Blue-tailed mole skink (*Eumeces egregius lividus*)

Listed Species - Birds

Mississippi sandhill crane (*Grus Canadensis pulla*)
Bald eagle (*Haliaeetus leucocephalus*)
Florida scrub jay (*Aphelocoma coerulescens coerulescens*)
Red-cockaded woodpecker (*Picoides borealis*)

Listed Species - Mammals

Florida panther (*Felis concolor coryi*)

Candidate Species - Plants

Incised groovebur (*Agrimonia incisa*)

Carolina lead-plant (*Amorpha georgiana* var. *confusa*)
 Georgia lead-plant (*Amorpha georgiana* var. *georgiana*)
 Southern three-awned grass (*Aristida simpliciflora*)
 Southern milkweed (*Asclepias viridula*)
 Chapman's aster (*Aster chapmani*)
 Coyote-thistle aster (*Aster eryngiifolius*)
 Pine-woods aster (*Aster spinulosus*)
 Sandhills milk-vetch (*Astragalus michauxii*)
 Purple balduina (*Balduina atropurpurea*)
 Hairy wild-indigo (*Baptisia calycosa* var. *villosa*)
 Scare-weed (*Baptisia simplicifolia*)
 Ashe's savory (*Calamintha ashei*)
 Sand grass (*Calamovilfa curtissii*)
 Piedmont jointgrass (*Coelorachis tuberculosa*)
 Large-flowered rosemary (*Conradina grandiflora*)
 Tropical waxweed (*Cuphia aspera*)
 Umbrella sedge (*Cyperus grayoides*)
 Dwarf burhead (*Echinodorus parvulus*)
 Telephus spurge (*Euphorbia telephioides*)
 Wiregrass gentian (*Gentiana pennelliana*)
 Florida beardgrass (*Gymnopogon floridanus*)
 Hartwrightia (*Hartwrightia floridana*)
 Mock pennyroyal (*Hedeoma graveolens*)
 Spider-lily (*Hymenocallis henryae*)
 Thick-leaved water-willow (*Justicia crassifolia*)
 White-wicky (*Kalmia cuneata*)
 Tiny bog buttons (*Lachnocaulon digynum*)
 Pine pinweed (*Lechea divaricata*)
 Godfrey's blazing star (*Liatris provincialis*)
 Slender gay-feather (*Liatris tenuis*)
 Panhandle lily (*Lilium iridollae*)
 Bog spicebush (*Lindera subcoriacea*)
 Large-fruited flax (*Linum macrocarpum*)
 Harper's grooved-yellow flax (*Linum sulcatum* var. *harperi*)
 West's flax (*Linum westii*)
 Boykin's lobelia (*Lobelia boykinii*)
 White birds-in-a-nest (*Macbridea alba*)
 Carolina bogmint (*Macbridea caroliniana*)
 Southern marshallia (*Marshallia ramosa*)
 Bog asphodel (*Narthecium americanum*)
 Fall-flowering ixia (*Nemastylis floridana*)
 Florida bear-grass (*Nolina atopocarpa*)
 Savanna cowbane (*Oxypolis ternata*)
 Naked-stemmed panic grass (*Panicum nudicaule*)
 Carolina grass-of-parnassus (*Parnassia caroliniana*)
 Wavyleafwild quinine (*Parthenium radfordii*)

Chapman's butterwort (*Pinguicula planifolia*)
 Bent golden-aster (*Pityopsis flexuosa*)
 Pineland plantain (*Plantago sparsiflora*)
 Wild coco, eulophia (*Pteroglossaspis ecristata*)
 Sandhills pixie-moss (*Pyxidantha barbulata* var. *brevifolia*)
 St. John's Susan, yellow coneflower (*Rudbeckia nitida* var. *nitida*)
 Bog coneflower (*Rudbeckia scabrifolia*)
 White-top pitcherplant (*Sarracenia leucophylla*)
 Wherry's pitcherplant (*Sarracenia rubra* ssp. *wherryi*)
 Florida skullcap (*Scutellaria floridana*)
 Scarlet catchfly (*Silene subciliata*)
 Carolina goldenrod (*Solidago pulchra*)
 Spring-flowering goldenrod (*Solidago verna*)
 Wireleaf dropseed (*Sporobolus teretifolius*)
 Pickering's morning-glory (*Stylisma pickeringii*)
 Pinel and hoary-pea (*Tephrosia mohrii*)
 Smooth bog-asphodel (*Tofieldia glabra*)
 Shinner's false-foxglove (*Tomanthera (Agalinis) pseudaphylla*)
 Least trillium (*Trillium pusillum* (5 varieties))
 Chapman's crownbeard (*Verbesina chapmanii*)
 Variable-leafcrownbeard (*Verbesina heterophylla*)
 Drummond's yellow-eyed grass (*Xyris drummondii*)
 Harper's yellow-eyed grass (*Xyris scabrifolia*)

Candidate Species - Insects

Buchholz's dart moth (*Agrotis buchholzi*)
 Aphodius tortoise commensal scarab beetle (*Aphodius troglodytes*)
 Arogos skipper (*Atrytone arogos arogos*)
 Copris tortoise commensal scarab beetle (*Copris gopheri*)
 Sandhills clubtail dragonfly (*Gomphusparvidens carolinus*)
 Spiny Florida sandhill scarab beetle (*Gronocarus multispinosus*)
 Prairie mole cricket (*Gryllotalpa major*)
 Mitchell's satyr (*Neonympha mitchellii francisci*)
 Onthophagus tortoise commensal scarab beetle (*Onthophagus polyphemi*)
 Carter's noctuid moth (*Spartiniphaga carterae*)

Candidate Species - Amphibians

Flatwoods salamander (*Ambystoma cingulatum*)
 Gopher frog (*Rana areolata*)
 Carolina gopher frog (*Rana capito capito*)
 Dusky gopher frog (*Rana capito sevosa*).

Candidate Species - Reptiles

Gopher tortoise (*Gopherus polyphemus*)
Florida scrub lizard (*Sceloporus woodi*)
Southern hognose snake (*Heterodon simus*)
Black pine snake (*Pituophis melanoleucus lodingi*)
Northern pine snake (*Pituophis melanoleucus melanoleucus*)
Florida pine snake (*Pituophis melanoleucus mugitus*)
Short-tailed snake (*Stilosoma extenuatum*)

Candidate Species - Birds

Southeastern American kestrel (*Falco sparveriuspaulus*)
Loggerhead shrike (*Lanius ludovicianus*)
Bachman's sparrow (*Aimophila aestivalis*)
Henslow's sparrow (*Ammodramus henslowii*)

Candidate Species - Mammals

Florida weasel (*Mustela frenata peninsulae*)
Florida black bear (*Ursus americanus floridanus*)
Florida mouse (*Podomys floridanus*)
Sherman's fox squirrel (*Sciurus niger shermani*)

APPENDIX 2. ANNOTATED LIST OF ANT SPECIES COLLECTED

***Hypoponera opaciceps* (Mayr): (Ponerinae)**

Distribution: Southeastern/Eastern

Considered a tramp species by some collectors, *Hypoponera opaciceps* prefers moist, disturbed habitats where it nests in soil and rotten wood (Creighton 1950, Deyrup et al. 1988, Taylor 1967).

***Hypoponera opacior* (Forel): (Ponerinae)**

Distribution: Southeastern

These ants are widespread throughout the South in mesic and/or xeric forests. Like others in this genus, *opacior* nests in soil and rotten wood (Creighton 1950, Deyrup et al. 1988, Taylor 1967).

***Ponera pennsylvanica* Buckley: (Ponerinae)**

Distribution: Eastern

This is one of only two native species of *Ponera* in the U.S. They are widely distributed in the eastern U.S. *Ponera pennsylvanica* prefer forests, but are found in open and conifer habitats in the Southeast. Colonies are very small, less than 30 workers, and are often found in litter, rotten wood, and under stones. This species is insectivorous foraging in and on the soil surface (Creighton 1950, MacKay and Anderson 1991, Taylor 1967).

***Pseudomyrmex pallida* Smith: (Pseudomyrmecinae)**

Distribution: Southern

There are only nine species of this genus in the U.S., but identification can be difficult because of geographic variation within a species. *Pseudomyrmex pallida* ranges

from North Carolina to Florida and over to southern California. This species is typically found nesting in grassy fields and marshes along coastal areas. In Florida it is often collected from *Andropogon*. Colonies are polydomous and may be monogyne or polygyne (Carter 1962, Ward 1985).

***Aphaenogaster carolinensis* Wheeler: (Myrmecinae)**

Distribution: Southeastern

This species is associated with wet, wooded sites; everything from pine forests to bayheads to swamps, but rarely in open fields or unwooded sites. I collected this species from a plot that was adjacent to bayhead forest. It was only collected prior to the first applied fire. *Aphaenogaster carolinensis* nest in rotten logs, fallen pinecones, litter or the upper layer of soil. They forage within litter for other arthropods and vegetation, particularly fungi. Other ant species that are frequently collected in the same habitat as *A. carolinensis* are *Hypoponera opacior* and *H. opaciceps*, *Pheidole dentata*, *Crematogaster ashmeadi* and *C. clara*, *Cyphomyrmex rimosus*, *Camponotus castaneus*, *Partrechina* spp., and *Trachymyrmex septentrionalis* (Carroll 1975, Carter 1962).

***Crematogaster ashmeadi* (Mayr): (Myrmecinae)**

Distribution: Southeastern

Mark Deyrup at Archbold Biological Station, FL believes my specimens may not be *ashmeadi* but a new species he is currently describing (Deyrup unpublished). Until his description is accepted my specimens will be labeled as *Crematogaster ashmeadi*. This species is arboreal, nesting in branches and stumps of longleaf pine, hardwoods, and shrubs throughout Southeastern U.S. It is commonly collected from pine forests along coastal regions. Each colony occupies a single tree. In areas where *Crematogaster*

ashmeadi is the dominant species, it serves as the primary food source of red-cockaded woodpeckers. Because this species is rather dominant it has been reported to outcompete *S. invicta* at a food source (Carter 1962, Hahn and Tschinkel 1997).

***Crematogaster missouriensis* (Smith): (Myrmecinae)**

Distribution: Central

This species is typically found in the central portion of the U.S. from Texas to Missouri (Creighton 1950).

***Crematogaster pilosa* Emery: (Myrmecinae)**

Distribution: Eastern

Crematogaster pilosa is common along the Atlantic and Gulf Coastal Plains.

Mainly arboreal, it prefers mesic, pine habitats to xeric uplands. This species builds its nests in rotten wood (Carter 1962, Johnson 1988).

***Cyphomyrmex rimosus* (Spinola): (Myrmecinae)**

Distribution: Introduced from Argentina, Brazil, and/or Venezuela

Cyphomyrmex rimosus spread throughout the Southeast after introduction into the U.S. Most species in this genus are from South America, and like other introduced species prefer disturbed, wet habitats (or mesic woods). The common name for *Cyphomyrmex rimosus* is larger little fungus ant. These ants have small colonies, usually less than 200 members and have 2 or 3 queens. This species is a fungus grower building its nest under leaves, fallen wood, grass clumps, etc. Their fungus gardens are grown on insect feces and body parts. These ants are slow moving and play dead when disturbed. Although fairly common, *Cyphomyrmex rimosus* is “probably of minimal concern as far

as impact on natives” since as a fungus grower, it doesn’t appear to be in competition with native species (Deyrup et al. 2000, Snelling and Longino 1992).

***Monomorium viride* Brown: (Myrmecinae)**

Distribution: Southeastern

There are few native species of this genus. *Monomorium viride* tends to prefer open areas with sandy soils for nest building. Colonies are polygynous (Creighton 1950, DuBois 1986).

***Myrmecina americana* Emery: (Myrmecinae)**

Distribution: Eastern

These ants construct small colonies of less than 100 individuals in soil or under leaf litter. They tend to prefer forests. *Myrmecina americana* are rather timid and play dead when disturbed. Species of this genus also inhabit the Indo-Australian region; there are only 2 or 3 species in U.S. This species is one of the most common ant species collected in North Carolina (Carter 1962).

***Pheidole dentata* Mayr: (Myrmecinae)**

Distribution: Southeastern

This species like other *Pheidole* are seed harvesters, but one observer referred to them as carnivores. They have dimorphic castes. *Pheidole dentata* nests in downed trees and soil. Much research has been done on their acute defense response to *S. invicta*. They are more common in coastal habitats with open grasslands, but can be found just about anywhere (Carter 1962, Creighton 1950, Naves 1985, Wilson 1975).

***Pheidole dentigula* Smith: (Myrmecinae)**

Distribution: Southeastern

This species is most commonly collected in mesic pine habitats along the coast. Similar to other *Pheidole*, this species has a dimorphic worker caste. They nest in soil, but prefer litter or rotten logs and are omnivorous eating everything from seeds to dead arthropods (Carter 1962, Creighton 1950, Naves 1985).

***Pheidole flavens* Roger: (Myrmecinae)**

Distribution: Introduced from Neotropics

Called the Yellow Big-Headed Ant, *Pheidole flavens* is found in disturbed habitats and mesic woodlands. They prefer drier habitats to wetter ones. Nests are in rotten wood. These ants are scavengers (Deyrup et al. 2000).

***Pheidole metallescens* Emery: (Myrmecinae)**

Distribution: Southeastern

Found primarily from the Gulf states this species was collected in open, pine sites. They eat small seeds and dead arthropods. *Pheidole metallescens* as the name implies, has a metallic sheen that is reflected when viewed under a microscope (Carter 1962, Creighton 1950, Naves 1985).

***Smithistruma margaritae* (Forel): (Myrmecinae)**

Distribution: Introduced from the Neotropics

The opaque moustache ant is rare in the U.S. (Deyrup et al. 2000).

***Smithistruma talpa* (Weber): (Myrmecinae)**

Distribution: Southeastern

Most species of *Smithistruma* occur in temperate zones unlike its close relative *Strumigenys*, which has its origins in the tropics. This species is common in the Southeast. *Smithistruma* are predators of Collembola. Colonies are small, about 300 workers, and are found under rocks, logs, leaf litter, etc. This genus was thought to occur only in mesic habitats, but availability, not habitat, dictates colony establishment. *Smithistruma talpa* has been collected in everything from *Andropogon* sod to water oak bottoms. Studying colonies of this species is often difficult because workers feign dead when disturbed. Because species in this genus are small, cryptic ants, samples often contain undescribed species (Brown 1953).

***Solenopsis carolinensis* Forel: (Myrmecinae)**

Distribution: Eastern

Called thief ants, members of this species nest in soil and are predators of other arthropods. These ants are very small, 2mm or less (Thompson 1989).

***Solenopsis invicta* Buren: (Myrmecinae)**

Distribution: Introduced from Brazil

The red imported fire ant is found in many habitats, but prefers open, disturbed habitats with high water tables. *Solenopsis invicta* are aggressive predators and scavengers. Fire ants that could be found at Lake Ramsay other than *S. invicta* include: *Solenopsis richteri*, black imported fire ant; *Solenopsis geminata*, tropical fire ant; and *Solenopsis xyloni*, southern fire ant (Davidson and Stone 1989, Deyrup et al. 2000). Only *S. invicta*, the red imported fire ant, was collected.

***Strumigenys louisianae* Roger: (Myrmecinae)**

Distribution: Southeastern

Strumigenys is the largest genus in the Dacetini. This species ranges from Tennessee to Argentina, where it is often collected from coastal areas. *Strumigenys louisianae* is the only species of this genus in the reartic region. The type locality, as the name implies, is Louisiana. There is very little variation within the species among U.S. collections; variation increases towards Central and South America. These are slow moving ants from small colonies. They feed on Collembola and will remove them from other ant species' colonies. Creighton (1950) writes, "When one has observed the great deliberation with which *Strumigenys* moves, it seems remarkable that they should have adapted themselves to a diet consisting of insects so much more active than themselves." Jaws of these ants are held open and snap shut when Collembola contact trigger hairs on maxillary lobes. Nests are constructed in rotten wood, pine duff, under bark, etc. They often build nests close to or live within other ant colonies, such as *Aphaenogaster fulva* (which was also collected from Lake Ramsay WMA) (Carter 1962, Creighton 1950).

***Trachymyrmex septentrionalis* (McCook): (Myrmecinae)**

Distribution: Eastern

This species can be found from Texas to New York. *Trachymyrmex* is a genus of fungus growing ants that form small (24 members), unaggressive (i.e. slow moving, feign death) colonies. Nests are built in hard packed soil or sand (typical soil at Lake Ramsay WMA) in open, grassy areas. Fungus gardens are attached to rootlets of grasses and shrubs. Caterpillar feces are added to root tissues to enhance fungal growth. Colonies

have multiple queens, but polygyny does not increase colony size (Carter 1962, Creighton 1950).

***Trichoscapa membranifera* (Emery): (Myrmecinae)**

Distribution: Introduced from Old World Tropics

The bare pygmy snapping ant is common in open pastures. They eat Collembola and Campodeidae (Deyrup et al. 2000).

***Tapinoma sessile* (Say): (Dolichoderinae)**

Distribution: U.S.

Tapinoma sessile can be found throughout the U.S. with no preference for habitat or nest site location, although their presence tends to be scarce in Gulf coast states.

Colonies consist of 2000-5000 individuals. They are omnivorous, but preferring sweets, I collected them at honey baits (Carter 1962, Creighton 1950).

***Brachymyrmex depilis* Emery: (Formicinae)**

Distribution: Eastern

These minute ants (1-2 mm) are difficult to identify to species. Creighton offers three pages on the taxonomic uncertainty of species' descriptions within this genus.

Brachymyrmex depilis was the most abundant ant species collected from soil in a coastal deciduous forest in Maryland, but was rarely collected from leaf litter. This species was also observed tending scale insects in their nests along the bases of salt marsh plants in Mexico. There does not appear to be a preferred habitat for this species since they have been collected from mountain and coastal areas in grasslands and forests. They nest in soil, tree stumps, rotten wood, etc. (Carter 1962, Creighton 1950, Lynch and Johnson 1988, Yensen et al. 1980).

***Brachymyrmex musculus* Forel: (Formicinae)**

Distribution: Introduced from Central America

The first report of this exotic ant in the U.S. was from Louisiana in 1978. The little mouse rover ant is common, especially in disturbed areas and pine habitats. They nest under bark and at the base of pines, and are known to eat honeydew. Not much more is known about *Brachymyrmex musculus* and its classification remains problematic (Deyrup et al. 2000).

***Camponotus castanea* (Latreille): (Formicinae)**

Distribution: Eastern

Although this genus is referred to as carpenter ants, this particular species nests in soil. I observed them nesting in abandoned *S. invicta* mounds. This species is common in the coastal plain of North Carolina. Habitats include longleaf pine savannas and mixed pine and oak forests (Carter 1962, Creighton 1950).

***Formica pallidefulva* Latreille: (Formicinae)**

Distribution: Eastern

This species establishes small colonies in grass tufts or under debris. They are often victims of slave-making ants such as *Polyergus lucidus*. This species is found in both mesic and xeric habitats (Carter 1962).

***Paratrechina faisonensis* (Forel): (Formicinae)**

Distribution: Southeastern

This species commonly occurs in mesic woodlands in all states east of the Mississippi and south of New Jersey. Apparently they are uncommon in a longleaf pine

savanna although my sites were surrounded by bayhead forest. *Paratrechina concinna* is found in grasslands and marshes. *Paratrechina arenivega*, which prefer fire disturbed habitats, would have been more likely to be collected from a longleaf pine savanna. *Paratrechina faisonensis* nests in rotten branches, tree roots, and litter. This species tends to be a dominant species in its natural habitat along with *Formica pallidefulva*, *Lasius alienus*, *Aphaenogaster rudis*, or *Prenolepis imparis*. As a genus they are known to feed on aphid honey-dew, nectar and other insects (Creighton 1950, Trager 1984).

***Polygerus lucidus* Mayr: (Formicinae)**

Distribution: Eastern

Polygerus lucidus are slave-making ants; founding queens take over a host nest. They depend on slaves to forage and feed the colony. These ants have been collected from grassy pine areas of the North Carolina piedmont and coastal plain (Carter 1962, Creighton 1950).

VITA

Deanna Colby was born in Kenmore, New York, on February 8, 1968. Her family moved to Johnson City, Tennessee, during her early teens and have called it home ever since. She completed both her Bachelor of Science degree in biology and Master of Education at Milligan College, Tennessee. She received her Master of Science degree in entomology and plant pathology from The University of Tennessee. She began her doctoral studies in the Department of Entomology at Louisiana State University under the guidance of Dr. Dorothy Prowell, Austin C. Thompson Endowed Professor. Her research was made possible by a United States Environmental Protection Agency STAR graduate research fellowship. She graduated in August, 2002.