EFFECTS OF HABITAT TYPE AND DISTANCE FROM EDGE ON ANT

COMMUNITY COMPOSITION IN A DEGRADED NATURAL AREA

Matthew D. Trager [1,2]* and Ian J. Fiske[2]

[1] School of Natural Resources and Environment, University of Florida

Gainesville, FL 32611 USA

[2] Department of Wildlife Ecology and Conservation, University of Florida

Gainesville, FL 32611 USA

* Author for correspondence:

Department of Wildlife Ecology and Conservation

University of Florida

PO Box 110430

Gainesville, FL 32611-0430

USA

E-mail: mtrager@ufl.edu; Phone: 352-846-0527; Fax: 352-392-6984

Word count: 2976 words, 46 references, 2 tables, 4 figures

Abstract

We examined the effects of habitat type and distance from the edge on ant species richness and community composition and we assessed the ability of species to dominate experimental bait stations in degraded longleaf pine savanna and oak hammock in northern Florida. We were particularly interested in the relationships between non-indigenous species, including *Solenopsis invicta*, and the native ant community. Species richness per pitfall trap was significantly higher in the pine habitat, but distance from the edge had no effect in either habitat. Although NMDS ordination showed substantial separation in community composition between the two habitat types, it also did not show any effect of distance from edge. Non-indigenous ant species, of which *S. invicta* was the most common, were significantly more abundant in the pine habitat. There was no relationship between the number of introduced and native species, but in the pine habitat the presence of *S. invicta* was significantly associated with lower richness of native ant species at small spatial scales. A small subset of the species dominated the bait stations, led by *S. invicta*. The results of the baiting study suggest that at least some non-indigenous species may persist in diverse ant communities due to their ability to find and dominate food resources.

Introduction

Ants (Hymenoptera: Formicidae) are diverse, abundant and important

components of most tropical and temperate terrestrial systems (Hölldobler and Wilson

1990). The composition of ant communities may be influenced by interspecific

competition (Davidson 1977, Andersen and Patel 1994, Gibb 2005), variation in resource

availability and habitat quality (Palmer 2003, Boulton et al. 2005, Dauber et al. 2005),

and temporal changes in activity (Herbers 1985, Bestelmeyer 2000, Albrecht and Gotelli

2001). Research in a wide variety of systems has shown that the short- and long-term

effects of human activities affect ant communities of disturbed or developed areas

(Abensperg-Traun et al. 1996, Bestelmeyer and Wiens 1996, Vasconcelos 1999).

Because they are so sensitive to habitat variation and respond quickly to changes in

habitat quality, ants may be excellent indicators of land management practices and

restoration efforts (Majer 1996, Andersen et al. 2002, Andersen and Majer 2004,

Ottonetti et al. 2006). Both habitat patch size and edge effects have been shown to affect

the presence and relative abundance of ants in some systems (Punttila 1996, Abensperg-

Traun and Smith 1999, Bolger et al. 2000, Gibb and Hochuli 2002, Braschler and Baur

2003, Bruhl et al. 2003, Dauber and Wolters 2004). However, it is important to note that

the exact effects of anthropogenic habitat alteration vary among systems and among

species within systems (Kotze and Samways 2001).

The state of Florida (USA), with 218 species, has exceptionally high ant diversity

for the temperate zone (Deyrup 2003). This fauna includes at least 52 non-indigenous

species (Deyrup et al. 2000), some of which are known to be highly competitive and have

negative effects on populations of native ants and other arthropods (Porter and Savignano

1990, Suarez et al. 1998, Deyrup et al. 2000, Gotelli and Arnett 2000, Zettler et al. 2004). At large scales the abundance of native and exotic species may be positively correlated due to external factors such as productivity and habitat heterogeneity (Morrison and Porter 2003, Davies et al. 2005, Harrison et al. 2006), but the strong competitive ability of the most successful invasive ant species often limits coexistence at small scales (Punttila et al. 1996, Morrison 2000). Because the invasion of non-indigenous species is often facilitated by anthropogenic disturbance, degraded areas often have depauperate native ant communities comprising many non-native species (Tschinkel 1988, Stiles and Jones 1998, Suarez et al. 1998, Zettler et al. 2004).

We studied the ant communities of oak hammock and longleaf pine savanna in a degraded natural area on the University of Florida's Gainesville campus. The site is small and highly fragmented, making it ideal for investigating the effects of small-scale habitat variation and the presence of invasive, non-indigenous ant species on overall ant community composition. We had three primary objectives in conducting this study: 1) to identify relationships between habitat type and community composition, 2) to determine the effects of distance from edge on ant community characteristics and 3) to assess the competitive ability of the most abundant ant species at the site using baits. In addressing these questions we were particularly interested in the how non-indigenous ant species, particularly the red imported fire ant (*Solenopsis invicta*), responded to habitat and edge effects and how their diversity compared with that of native species.

Methods

Study site

We conducted this study in the University of Florida's Natural Area Teaching Lab (NATL), a 24 ha site comprising a variety of upland and wetland habitats typical of northern peninsular Florida. We sampled ants in two small tracts of longleaf pine (*Pinus palustris*) that had been recently burned as part of an ongoing restoration project and an adjacent tract of oak hammock. There is evidence that single fires can alter the short-term community composition of ants (Izhaki et al. 2003) and that ant assemblages differ within a site according to time since fire (Lubertazzi 2003). All sites were grazed within the past 100 years and as recently as 40 years ago the two pine sites and one section of the hardwood sites were open-grown longleaf savannah. The study areas were separated by dirt roads that contrasted sharply with the bordering vegetation and provided habitat for a number of habitat generalist ants and open-ground specialists that were otherwise rare at the site (MDT, unpublished data).

Collection methods

We collected ants with pitfall traps and baits, methods that are complementary for generating robust species lists in the region (King and Porter 2005). Pitfall traps were 15 dram vials (approx. 3 cm diameter) partially filled with propylene glycol and left open for 48 hr. We arranged the pitfall traps in transects (6 traps per transect) running 15 m from the habitat edge perpendicularly into the study area. This allowed both a thorough sampling of the ant community at the site and an assessment of variation in community composition due to habitat preference or edge effects. We sampled two contiguous but distinct areas of the hardwood hammock and the two tracts of longleaf pine in late May and again in July 2005. For each sample period we placed 30 pitfall traps arranged in five transects in the two areas of each habitat type, totaling 240 pitfall traps for the study.

Bait stations were 15 dram vials within which we placed a small piece of Vienna

sausage. We placed the vials on their sides and opened them for one hour, after which

we capped the vials and then identified and counted all ants present at the bait in the lab.

The sampling design was identical to that used for the pitfall traps but we used different

transects and sampled only once in mid-June 2005 for a total of 120 bait stations.

Data analysis

The pitfall trap data were used to generate species lists to describe the ant

community at the site, whereas the bait data were used primarily to demonstrate the

competitive ability of individual species relative to their abundance. Because pitfall traps

within transect were not independent samples, we used linear mixed-effects models that

treated individual traps as repeated measures within transects to determine the effects of

habitat type, sampling date and number of exotic species on species richness. We

followed a standard protocol for linear mixed effects models (Pinheiro and Bates 2002)

using maximum likelihood methods to fit model parameters. The two habitats were

analyzed separately to test for potential interactions between native and introduced

species because they differed substantially in the presence and abundance of non-

indigenous species. Differences in community composition due to habitat preference,

distance from edge and sampling date were assessed with non-metric multidimensional

scaling (NMDS). We used logistic regression to determine the effects of sampling

method on the capture probability for each of the five most abundant species (*Solenopsis

invicta*, *S. carolinensis*, *Pheidole dentata*, *P. dentigula* and *P. moerens*). Only pitfall

traps or bait stations that contained ants were included in this analysis. Data analysis was

conducted with R v. 2.2.1 (R Development Core Team 2005). All species identifications

were made by the authors and voucher specimens have been deposited at the Florida

State Collection of Arthropods (Florida Department of Agriculture, Division of Plant

Industry, Gainesville, FL).

Results

Pitfall traps

In the pitfall traps we collected 2550 individual ants from 28 species, 7 of which

are not native to Florida (Table 1). Abundance of species varied greatly, ranging from

882 individuals in 156 traps for *Solenopsis carolinensis* to one individual in one trap for

Aphaenogaster floridana, *Eurhopalothrix floridana* and an unidentified *Pyramica*. Total

species richness was higher in the longleaf pine habitat during both sampling periods (for

both May and July: pine = 22 species, oak = 16 species). Date of sampling did not

significantly affect species richness per trap, but there was significant variation between

the two habitats (Likelihood Ratio = 17.47, $P < 0.0001$). Species richness per trap was

approximately 50% higher and more variable in the pine habitat (mean $\pm$ SE = 3.86 $\pm$

0.30) than in the oak hammock (mean $\pm$ SE = 2.46 $\pm$ 0.21; Fig. 1). Although there was a

marginally significant effect of distance from the edge on species richness per trap

(Likelihood Ratio = 3.42, $P = 0.064$), there was no clear pattern indicating a directional

trend (Fig. 2). Maximum likelihood tests found no significant interaction between habitat

type and distance from the edge on species richness per trap so it was not included in the

final model. Community composition varied distinctly between habitats and between

sampling periods (Fig. 3). However, there was no evidence that distance from the edge

affected community composition in a consistent manner.

There was no significant difference between habitats in species richness per trap of native ants (Likelihood Ratio = 0.071, P = 0.79). However, there were significantly more individuals of non-indigenous species per trap in the longleaf pine habitat (mean ± SE = 4.58 ± 0.82) than in the oak hammock (mean ± SE = 0.16 ± 0.58; $F_{1,38}$ = 29.37, P < 0.0001). Distance to edge did not significantly improve model fit (Likelihood Ratio = 1.77, P = 0.18), indicating that there was no edge effect on the number of individuals of non-indigenous species in this system (Fig. 4). Furthermore, the longleaf pine habitat had more non-indigenous ant species than the oak hammock both absolutely (pine = 6, oak = 2) and as a proportion of the total species richness (pine = 0.24, oak = 0.10). This important difference in response between native and non-indigenous species certainly contributed to the separation of the communities of the pine and oak habitats in the ordination plot (Fig. 3).

There was no overall effect of the number of non-indigenous ant species or the number of non-indigenous individuals on the number of native species per trap (Likelihood Ratio = 0.76, P = 0.38 and Likelihood Ratio = 0.91, P = 0.34, respectively). Nor was there any effect of the number of introduced species on native species richness per trap when the two habitats were analyzes separately (Pine: t = 0.52, df = 98, P = 0.60; Oak: t = 1.35, df = 98, P = 0.18). In the pine habitat, the presence of *Solenopsis invicta* was not related to the number of individuals of native species but was significantly associated with lower native species richness at the trap level (t = 0.12, df = 98, P = 0.90 and t = -2.11, df = 98, P = 0.037, respectively). In contrast, in the oak habitat, the presence of *Solenopsis invicta* was not related to either the number of individuals or

richness of native species per trap (t = -0.70, df = 96, P = 0.48 and t = 0.41, df = 96, P = 0.68).

Bait stations

There were 14 species present at the baits, totaling 5535 individuals (Table 2). One species at the bait stations, *Monomorium viride*, was not collected in the pitfall traps. Of the 117 bait stations recovered, 88 contained ants and only a single ant species was present at approximately three-quarters (n = 65) of these baits. *Solenopsis invicta*, *S. carolinensis* and *Pheidole moerens* dominated the most baits (19, 18 and 9, respectively). Logistic regression showed that the probability of capture of *S. invicta* did not differ between pitfall traps and baits, although there was a non-significant positive effect of the bait method on capture probability and the mean number of individuals per sample was much higher for bait stations. The three most abundant native species, *Solenopsis carolinensis*, *P. dentata* and *P. dentigula*, were all significantly less likely to be captured at the bait stations compared to the pitfall traps (p < 0.001 for all three species). The capture probability for *P. moerens* was also lower with the baits (p < 0.001), but this species was significantly more likely to be captured at the bait stations than in the pitfall traps in the oak hammock (p < 0.001).

Discussion

Habitat and resource partitioning in ant communities is determined by both environmental factors and interactions among potentially coexisting species (Hölldobler and Wilson 1990, Albrecht and Gotelli 2001). We did not analyze the habitat preferences of individual species, but there was clear separation between the ant communities of the

two habitat types and the affinity of species for either the oak hammock or degraded longleaf pine savannah is evident in Table 1. Additionally, we found more species in the pine habitat overall and significantly more species per pitfall trap in the longleaf pine tracts than in the oak hammock. Some species showed predictable patterns of abundance, such as the open habitat specialist *Dorymyrmex bureni*, but the occurrence of other species was more enigmatic, such as the higher abundance of the litter-dwelling *Pyramica* and *Strumigenys* species in the recently burned pine habitat than in the oak hammock. Although there is some evidence that ant communities change with distance from habitat edge (Bolger et al. 2000, Dauber and Wolters 2004, Ness 2004), we detected no significant edge effects on species richness or community composition. It is possible that we failed to capture the edge effect; alternatively, it is possible that the degradation of the habitat renders even interior portions as functionally marginal habitat for ants.

Our analyses failed to show any correlation between introduced species as a whole and the number of native species present at the trap level. However, it appeared that the higher species richness in the longleaf pine tracts at both the trap and habitat level was entirely attributable to the presence of more non-indigenous species. Previous research has shown that disturbance can increase the abundance of many introduced ant species sometimes with consequent negative effects on the native ant community (Suarez et al. 1998, Bolger et al. 2000, Andersen and Majer 2004). Both the oak hammock and the longleaf pine area have been substantially degraded, but the longleaf pine habitat currently experiences more frequent and severe disturbances (e.g., fire, woody vegetation removal and soil perturbation from planting pines and wiregrass) as part of ongoing ecological restoration work. It remains a possibility that some native species have been

displaced in these sites due to competition with introduced species, but our study was not designed to test this hypothesis directly.

The red imported fire ant, *Solenopsis invicta*, was rare in the oak hammock (13 individuals in 4 pitfall traps), but was among the most abundant species in the longleaf pine savanna (279 individuals in 70 pitfall traps). This species was also the most abundant ant at baits, both in number of individuals and number of baits dominated. The presence of *S. invicta* can reduce native ant diversity at both small (Porter and Savignano 1990, Morris and Steigman 1993) and large spatial scales (Gotelli and Arnett 2000). However, there is also evidence suggesting that the initial effects of fire ant invasion may eventually relax and that other habitat factors may promote both native ant diversity and fire ant abundance (Morrison 2002, Morrison and Porter 2003). We found that the presence of *S. invicta* was negatively associated with native ant species richness at the trap level in the longleaf pine savanna but not in the oak hammock or when data from both habitats were pooled. This suggests that at very small scales (i.e., the trap level), *S. invicta* outcompeted or reduced the species density of native ants, and perhaps other non-indigenous ant species as well. Such a finding is consistent with the widely-recognized ability of *S. invicta* to dominate resources and exclude other species (Deyrup et al. 2000, Morrison 2000) and is consistent with our data from the baiting study. Because *S. invicta* is abundant in both disturbed areas and frequently-burned longleaf pine savannas (Tschinkel 1988, Lubertazzi 2003, Zettler et al. 2004)) this species will likely persist at the site with unknown consequences for other ant species.

The ants captured at the bait stations included a smaller proportion of non-indigenous species than those captured with pitfall traps (14.3% and 25%, respectively).

However, these two species, *S. invicta* and *P. moerens* were among the three most dominant species at the baits and *S. invicta* was the only one of the five most abundant species overall that was not less likely to be captured with baiting. Similarly, although P. moerens was relatively rarely captured in the oak hammock with pitfall traps, it was the second most frequently captured species at the bait stations in that habitat. This finding suggests that these two species are dominant at the NATL site both numerically and competitively. The dominance of *S. invicta* in disturbed habitats in the southeastern US is widely recognized, and *P. moerens* has been identified as a potential threat to native ants or other arthropods in Florida (Deyrup et al. 2000). Our study confirms the former finding and indicates that the latter suggestion may merit further study.

In summary, we found strong support for the division of two distinct ant communities associated with different habitats in a degraded natural area in northern Florida but no effect of distance from the edge on community composition or species richness. Non-indigenous species were abundant in the longleaf pine habitat but relatively rare in the oak hammock, perhaps dues to differences in site history and current disturbance regime associated with ecological restoration efforts. We found some evidence of competitive exclusion by imported red fire ants at the trap level in the pine habitat, but no effect of *S. invicta* presence or abundance on native ants at larger scales. Although our study was conducted at a single site, the NATL is perhaps representative of many small, fragmented and degraded parks and natural areas in the region.

Acknowledgements

References

Abensperg-Traun, M., and G. T. Smith. 1999. How small is too small for small animals? Four terrestrial arthropod species in different-sized remnant woodlands in agricultural Western Australia. Biodiversity and Conservation **8**:709-726.

Abensperg-Traun, M., G. T. Smith, G. W. Arnold, and D. E. Steven. 1996. The effects of habitat fragmentation and livestock-grazing on animal communities in remnants of gimlet *Eucalyptus salubris* woodland in the Western Australian wheatbelt .1. Arthropods. Journal of Applied Ecology **33**:1281-1301.

Albrecht, M., and N. J. Gotelli. 2001. Spatial and temporal niche partitioning in grassland ants. Oecologia **126**:134-141.

Andersen, A. N., B. D. Hoffmann, W. J. Muller, and A. D. Griffiths. 2002. Using ants as bioindicators in land management: simplifying assessment of ant community responses. Journal of Applied Ecology **39**:8-17.

Andersen, A. N., and J. D. Majer. 2004. Ants show the way Down Under: invertebrates as bioindicators in land management. Frontiers in Ecology and the Environment **2**:291-298.

Andersen, A. N., and A. D. Patel. 1994. Meat ants as dominant members of Australian ant communities - an experimental test of their influence on the foraging success and forager abundance of other species. Oecologia **98**:15-24.

Bestelmeyer, B. T. 2000. The trade-off between thermal tolerance and behavioural dominance in a subtropical South American ant community. Journal of Animal Ecology **69**:998-1009.

Bestelmeyer, B. T., and J. A. Wiens. 1996. The effects of land use on the structure of ground-foraging ant communities in the Argentine Chaco. Ecological Applications **6**:1225-1240.

Bolger, D. T., A. V. Suarez, K. R. Crooks, S. A. Morrison, and T. J. Case. 2000. Arthropods in urban habitat fragments in southern California: Area, age, and edge effects. Ecological Applications **10**:1230-1248.

Boulton, A. M., K. F. Davies, and P. S. Ward. 2005. Species richness, abundance, and composition of ground-dwelling ants in northern California grasslands: Role of plants, soil, and grazing. Environmental Entomology **34**:96-104.

Braschler, B., and B. Baur. 2003. Effects of experimental small-scale grassland fragmentation on spatial distribution, density, and persistence of ant nests. Ecological Entomology **28**:651-658.

Bruhl, C. A., T. Eltz, and K. E. Linsenmair. 2003. Size does matter - effects of tropical rainforest fragmentation on the leaf litter ant community in Sabah, Malaysia. Biodiversity and Conservation **12**:1371-1389.

Dauber, J., T. Purtauf, A. Allspach, J. Frisch, K. Voigtlander, and V. Wolters. 2005. Local vs. landscape controls on diversity: a test using surface-dwelling soil

macroinvertebrates of differing mobility. Global Ecology and Biogeography **14**:213-221.

Dauber, J., and V. Wolters. 2004. Edge effects on ant community structure and species richness in an agricultural landscape. Biodiversity and Conservation **13**:901-915.

Davidson, D. W. 1977. Species diversity and community organization in desert seed-eating ants. Ecology **58**:711-724.

Davies, K. F., P. Chesson, S. Harrison, B. D. Inouye, B. A. Melbourne, and K. J. Rice. 2005. Spatial heterogeneity explains the scale dependence of the native-exotic diversity relationship. Ecology **86**:1602-1610.

Deyrup, M. 2003. An updated list of Florida ants (Hymenoptera : Formicidae). Florida Entomologist **86**:43-48.

Deyrup, M., L. Davis, and S. Cover. 2000. Exotic ants in Florida. Transactions of the American Entomological Society **126**:293-326.

Gibb, H. 2005. The effect of a dominant ant, *Iridomyrmex purpureus*, on resource use by ant assemblages depends on microhabitat and resource type. Austral Ecology **30**:856-867.

Gibb, H., and D. F. Hochuli. 2002. Habitat fragmentation in an urban environment: large and small fragments support different arthropod assemblages. Biological Conservation **106**:91-100.

Gotelli, N. J., and A. E. Arnett. 2000. Biogeographic effects of red fire ant invasion. Ecology Letters **3**:257-261.

Harrison, S., J. B. Grace, K. F. Davies, H. D. Safford, and J. H. Viers. 2006. Invasion in a diversity hotspot: Exotic cover and native richness in the Californian serpentine flora. Ecology **87**:695-703.

Herbers, J. M. 1985. Seasonal structuring of a north temperate ant community. Insectes Sociaux **32**:224-240.

Hölldobler, B., and E. O. Wilson. 1990. The ants. Belknap, Cambridge, UK.

Izhaki, I., D. J. Levey, and W. R. Silva. 2003. Effects of prescribed fire on an ant community in Florida pine savanna. Ecological Entomology **28**:439-448.

King, J. R., and S. D. Porter. 2005. Evaluation of sampling methods and species richness estimators for ants in upland ecosystems in Florida. Environmental Entomology **34**:1566-1578.

Kotze, D. J., and M. J. Samways. 2001. No general edge effects for invertebrates at Afromontane forest/grassland ecotones. Biodiversity and Conservation **10**:443-466.

Lubertazzi, D. a. W. R. T. 2003. Ant community change across a ground vegetation gradient in north Florida's longleaf pine flatwoods. Journal of Insect Science **3**:17pp.

Majer, J. D. 1996. Ant recolonization of rehabilitated bauxite mines at Trombetas, Para, Brazil. Journal of Tropical Ecology **12**:257-273.

Morris, J. R., and K. L. Steigman. 1993. Effects of polygyne fire ant invasion on native ants of a blackland prairie in Texas. Southwestern Naturalist **38**:136-140.

Morrison, L. W. 2000. Mechanisms of interspecific competition among an invasive and two native fire ants. Oikos **90**:238-252.

Morrison, L. W. 2002. Long-term impacts of an arthropod-community invasion by the imported fire ant, *Solenopsis invicta*. Ecology **83**:2337-2345.

Morrison, L. W., and S. D. Porter. 2003. Positive association between densities of the red imported fire ant, Solenopsis invicta (Hymenoptera : Formicidae), and generalized ant and arthropod diversity. Environmental Entomology **32**:548-554.

Ness, J. H. 2004. Forest edges and fire ants alter the seed shadow of an ant-dispersed plant. Oecologia **138**:448-454.

Ottonetti, L., L. Tucci, and G. Santini. 2006. Recolonization patterns of ants in a rehabilitated lignite mine in central Italy: Potential for the use of Mediterranean ants as indicators of restoration processes. Restoration Ecology **14**:60-66.

Palmer, T. M. 2003. Spatial habitat heterogeneity influences competition and coexistence in an African acacia ant guild. Ecology **84**:2843-2855.

Pinheiro, J. C., and D. M. Bates. 2002. Mixed effects models in S and S-Plus. Springer-Verlag, New York, USA.

Porter, S. D., and D. A. Savignano. 1990. Invasion of polygyne fire ants decimates native ants and disrupts arthropod community. Ecology **71**:2095-2106.

Punttila, P. 1996. Succession, forest fragmentation, and the distribution of wood ants. Oikos **75**:291-298.

Punttila, P., Y. Haila, and H. Tukia. 1996. Ant communities in taiga clearcuts: Habitat effects and species interactions. Ecography **19**:16-28.

R Development Core Team. 2005. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Stiles, J. H., and R. H. Jones. 1998. Distribution of the red imported fire ant, *Solenopsis invicta*, in road and powerline habitats. Landscape Ecology **13**:335-346.

Suarez, A. V., D. T. Bolger, and T. J. Case. 1998. Effects of fragmentation and invasion on native ant communities in coastal southern California. Ecology **79**:2041-2056.

Tschinkel, W. R. 1988. Distribution of the fire ants *Solenopsis invicta* and *S. geminata* (Hymenoptera, Formicidae) in northern Florida in relation to habitat and disturbance. Annals of the Entomological Society of America **81**:76-81.

Vasconcelos, H. L. 1999. Effects of forest disturbance on the structure of ground-foraging ant communities in central Amazonia. Biodiversity and Conservation **8**:409-420.

Zettler, J. A., M. D. Taylor, C. R. Allen, and T. P. Spira. 2004. Consequences of forest clear-cuts for native and nonindigenous ants (Hymenoptera : Formicidae). Annals of the Entomological Society of America **97**:513-518.

Table 1. Summarized data for the 28 species collected in pitfall traps, noting the number of individuals and the number of traps for each species, divided by habitat.

	Oak hammock		**Longleaf pine**	
Species	**Individuals**	**No. traps**	**Individuals**	**No. traps**
Aphaenogaster ashmeadi	3	3	14	10
Aphaenogaster carolinensis	7	7	0	0
Aphaenogaster floridana	1	1	0	0
Brachymyrmex depilis	3	3	7	4
Brachymyrmex musculus	0	0	40	13
Camponotus castaneus	27	19	11	9
Camponotus floridanus	22	20	8	5
Cardiocondyla emeryi	0	0	38	18
Crematogaster ashmeadi	1	1	1	1
Cyphomyrmex rimosus	0	0	23	15
Dorymyrmex bureni	0	0	56	14
Eurhopalothrix floridana	1	1	0	0
Formica pallidefulva	16	12	82	35
Hypoponera opacior	8	6	15	13
Odontomachus brunneus	47	37	31	23
Pheidole dentata	199	77	136	34
Pheidole dentigula	167	69	108	41
Pheidole metallescens	3	1	78	18
Pheidole moerens	6	4	188	38
Pyramica eggersi	0	0	8	6
Pyramica membranifera	0	0	8	2
Pyramica sp.	0	0	1	1
Solenopsis carolinensis	435	75	447	81
Solenopsis invicta	13	4	279	70
Solenopsis nickersoni	1	1	1	1
Solenopsis tennesseensis	1	1	3	3
Strumigenys louisianae	0	0	2	2
Trachymyrmex septentrionalis	2	2	2	2

Table 2. Summarized data for the 14 species collected at the bait stations, noting the number of individuals and the number of traps for each species, divided by habitat.

	Habitat			
	Oak hammock		**Longleaf pine**	
<u>**Species**</u>	<u>**Individuals**</u>	<u>**No. traps**</u>	<u>**Individuals**</u>	<u>**No. traps**</u>
Aphaenogaster ashmeadi	11	3	0	0
Aphaenogaster carolinensis	33	3	0	0
Camponotus floridanus	17	4	6	1
Cardiocondyla emeryi	0	0	1	1
Formica pallidefulva	3	3	19	1
Monomorium viride	0	0	1	1
Odontomachus brunneus	10	4	1	1
Paratrechina faisonensis	5	3	1	1
Pheidole dentata	19	5	89	6
Pheidole dentigula	24	4	0	0
Pheidole metallescens	61	1	0	0
Pheidole moerens	457	11	168	8
Solenopsis carolinensis	376	14	249	11
Solenopsis invicta	16	2	3968	26

Figure legends

Figure 1. Species richness per trap was significantly higher in the longleaf pine savanna than in the oak hammock. Boxplots show median (bold), minimum, maximum and interquartile range.

Figure 2. Species richness per trap was marginally significantly affected by distance from edge, but there was no clear directional trend. Boxplots show median (bold), minimum, maximum and interquartile range of all traps at each distance.

Figure 3. NMDS ordination plot of ant community composition. Traps in each habitat for each date were pooled according to distance from edge, and are labeled as such (1 = 1 m, 2 = 4 m, 3 = 7 m, etc.). There are clear differences between the two habitat types and the two sampling dates, but no apparent pattern in community composition related to distance from edge.

Figure 4. Number of individuals of all non-indigenous species per trap was significantly higher in the longleaf pine habitat but was not affected by distance from edge.
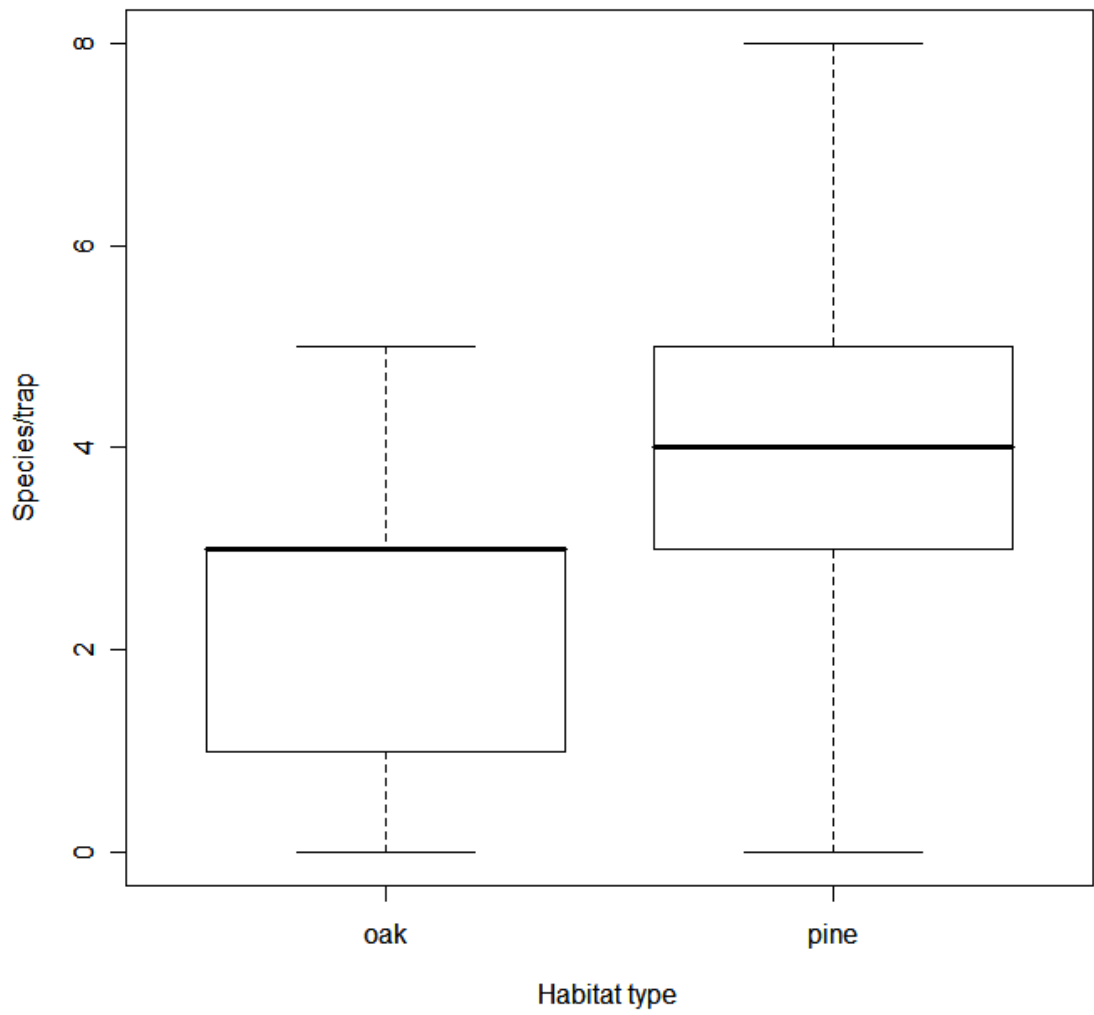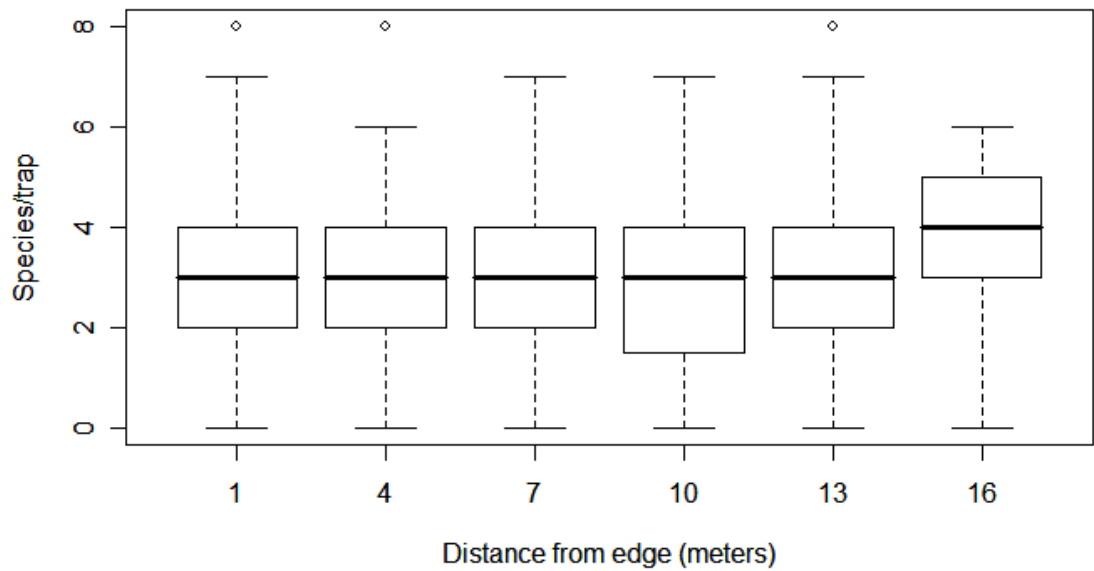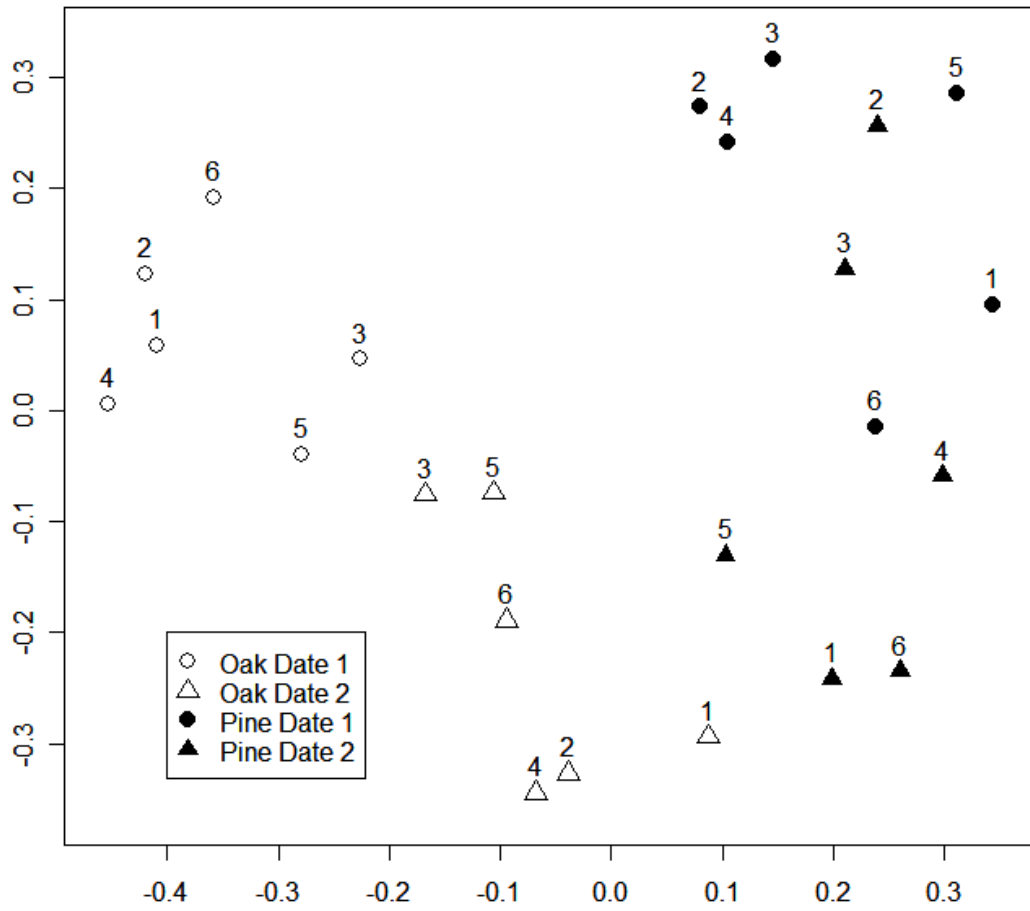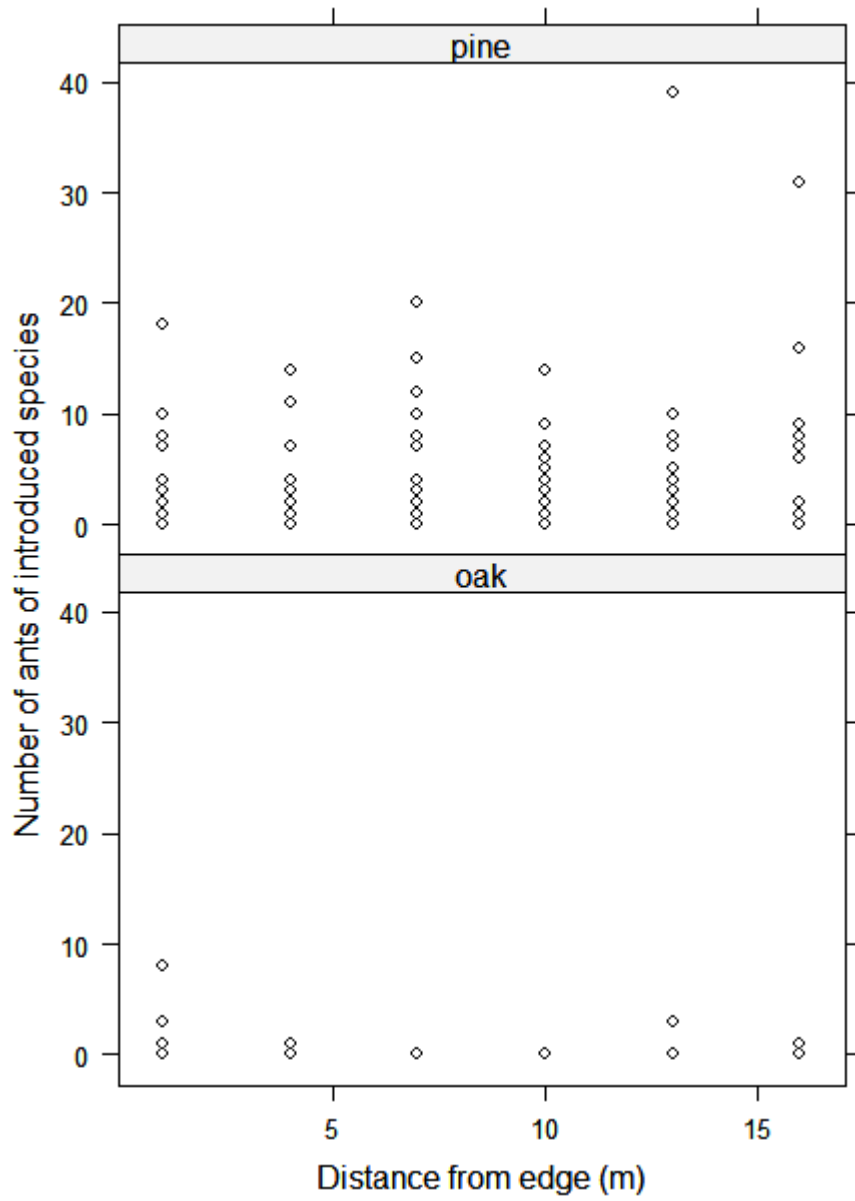
Fig. 1

Fig. 2

Fig. 3

Fig. 4