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The effects of prescribed fire on ant community composition in a temperate deciduous forest

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Abstract

Prescribed fire is a tool commonly used in land management to decrease wildfire frequency and promote plant diversity. However, the effects of prescribed fire on invertebrate communities, especially those within temperate deciduous forest, are poorly understood. I measured the response of epigeic ant communities in mixed mesophytic forest in Berea, Kentucky following prescribed burning. I used pitfall traps to repeatedly sample epigeic ants in replicate burned and unburned plots for up to 21 months postburn following two separate (2021 and 2022) prescribed fires. Ant species richness was similar between treatments (burn vs. control) and by burn year. Ant community composition generally differed between treatments and across years but was similar between the paired 2022 burned and unburned plots, probably due to the low intensity of that burn. The results of this study indicate that epigeic ant communities in an eastern deciduous forest are altered by prescribed burning, and do not return to normal activity levels after 1-year post-burn. Additional experimental studies are needed to determine the effects of fire intensity and frequency on ant assemblages in this setting.

Introduction

Ecological disturbance is any relatively discrete event in time that disrupts ecosystem, community, or population structure and alters resources or habitat (Pickett and White 1985). Fire is a natural disturbance in many terrestrial ecosystems and can vary widely in intensity and frequency (Swetnam and Betancourt 1990, Delcourt and Delcourt 1997). Fire in forested ecosystems often promotes plant biodiversity (Richter et al. 1982, Mitchell et al. 2006), which generates a bottom-up effect for organisms at higher trophic levels (Scherber et al. 2010). Historical suppression of wildfires in temperate forest of North America altered natural forest turnover (Ryan et al. 2013), leading to a decrease in biodiversity and habitat heterogeneity (Baker 1992, 1993). Consequently, intentional burning (i.e., “prescribed burning”) of forests has become increasingly common in forestry management operations to promote biodiversity, create habitat heterogeneity, and mitigate extreme wildfire events. (Agee and Skinner 2005, Finney et al. 2005, Prichard et al. 2010, Cochrane et al. 2012).

Fire regimes can vary in frequency (i.e., number of fires per unit time) and burn intensity (i.e., heat transfer per unit duration of fire, Hiers et al. 2009, Perry et al. 2011.). Most studies evaluating how different fire regimes influence forest components focus on plant communities (Pausas 1999, Franklin et al. 2001, Bond et al. 2004, Keeley 2006, Miller et al. 2019), but data on ecologically important groups, like epigeic (i.e., ground dwelling) arthropods, are lacking.

Ants are ecologically important components of forested ecosystems. They function as ecosystem engineers (Folgarait 1998, Jouquet et al. 2006), keystone seed dispersers (Lengyel et al. 2010, Gorb and Gorb 2013, Warren and Giladi 2014), and predators of many primary consumers (Folgarait 1998, Philpott and Armbrecht 2006). Fire can negatively impact ants directly through mortality and indirectly by reducing available nesting sites (Graham et. al 2009). Given that ants are numerically abundant, good bioindicators of disturbance (Anderson et al. 2002), and easily sampled, they may provide insight on how similar epigeic arthropods respond to prescribed fire in temperate deciduous forests.

Previous studies have examined the effect of prescribed fire on ant species richness and community composition in a variety of ecosystems, including prairies (Underwood and Christian 2009, Menke et al. 2015, Bonoan and McCarthy 2022), eucalypt forests (Andersen et al. 2009, Beaumont et al. 2012), and savannas (Izhaki et al. 2003, Houdeshell et al. 2011). Temperate deciduous forest cover 7.8 million km^2 worldwide and dominate most of the eastern United States (Allaby 2006). The few studies that were conducted in temperate deciduous forest habitats tended to examine only the short-term effects of prescribed burning on epigeic ant communities (Verble and Yanoviak 2013) or analyzed a general response of soil invertebrates to prescribed burning (Kalisz and Powell 2000).

Here, I investigate the effects of prescribed fire on the structure of epigeic ant assemblages (species composition and species richness) up to 1 year following a burn in an eastern deciduous forest in Kentucky, USA. I expected ant species richness to be lower in burned forest (Underwood and Quinn 2010, Verble and Yanoviak 2013). I also expected ant species composition in burned forest to be distinct from that of unburned forest (Beaumont et al. 2012).

Methods

Study Site

Field work was conducted at Berea College Forest, a 3642-hectare mixed mesophytic forest in Berea, Kentucky (37.5687° N, 84.2963° W). Berea Forest is managed in part via an annual prescribed burn program (Patterson and Singleton 2019). This study focused on the effects of two annual burns conducted in April (2021 and 2022). In each case, only one section of forest was burned, an adjacent unburned section of equal area was used as a control site and burn, and control sites did not overlap between years.

Experimental Design

Following each prescribed burn, I established three transects in the burned site and three similar transects in the adjacent control site. Each transect was 10 m long and separated from other transects by > 100 m. Hereafter, *treatment* refers to burned vs. unburned (control) forest and *year* refers to 2021 vs. 2022.

I used pitfall traps to sample epigeic ants along each transect. Each trap consisted of a 475 ml plastic cup (model No. S-22770, Uline, Pleasant Prairie, WI, USA), buried such that the cup opening was flush with the soil surface. A second cup was placed inside the buried cup to facilitate sample collection. Each buried cup assembly was sheltered with a roof of acrylic sheeting (ca. 0.6 x 15.2 x 15.2 cm) suspended 2.5 cm above the cup opening with landscape staples (Amagabeli, Beijing, China). A small amount of antifreeze (Model: RV and Marine antifreeze -50 °F, Splash, St. Paul, MN, USA) mixed with dish soap added to each trap served as the killing agent (Fig. 1).

In 2021, pitfalls traps were placed every meter on alternating sides of the transect for all six transects (Fig. 2). I placed 30 pitfall traps in burned forest and 30 pitfall traps in unburned forest, which were sampled weekly. In 2022, I sampled burned and unburned plots of forest following the 2022 burn along with the burned and unburned plots established in 2021. I reduced sampling intensity to 10 pitfalls (3, 3, and 4 traps for each transect) in each plot, resulting in a total of 40 pitfalls across 4 different plots in 2022 (Fig. 2), which were sampled monthly.

Statistical Analyses

Analyses include only trap data where at least one ant was present. I did not use raw abundance as a response due to the social nature of ants (Wilson and Hölldobler 2005). Instead, I used the total number of unique occurrences (i.e., the number of times a species was recorded in a pitfall trap through the entire sampling period.) The total number of unique occurrences for each species was pooled over time for each pitfall trap.

To determine whether the number of unique ant occurrences differed between treatments and year, I constructed a generalized linear mixed effect model (GLMM) with a negative binomial distribution to account for the extreme right-skew in the data. I included the interaction term between the main effects of treatment and year and included plot as a random effect (Table 1). I did not include days post-fire as an explanatory effect because pitfall data were pooled across samples within a year.

I used non-metric multidimensional scaling (NMDS) ordination to determine whether ant assemblages in burned forest were compositionally distinct from those in unburned forest (Fig. 3; PRIMER Software Version 7.0.21). I created a Bray-Curtis dissimilarity matrix using the number of unique occurrences for each species within each pitfall trap as the response. To determine whether forest status, year, and their interactions had a significant effect on ant communities, I used permutational multivariate analysis of variance (PERMANOVA; Table 2). I then ran a permutational test of dispersion (PERMDISP) to determine whether the potential differences were attributable to the main effects or to dispersion within the data (Table 3).

I conducted a post-hoc test to determine which treatment and year combinations were significantly different from each other (Table 4). I used Bonferroni adjusted p-values to account for multiple comparisons.

Results

I collected 4,947 ants comprised of 23 species and 13 genera over the 21 months of sampling. The total number of ants collected was distributed among 100 trap occurrences, for an average of 49.5 ants per trap collection over the course of the study. In most cases, each trap contained relatively few species. The most frequently collected species was *Aphaenogaster carolinensis* (n = 986 occurrences), and many species were represented by a single individual (e.g., *Camponotus castaneus*, *Formica pallidefulva*, *Proceratium chickasaw*, *Strumigenys ohioensis*, *Temnothorax curvispinosus*, and *Temnothorax schaumii*; Fig. 4)

Most species were collected in both burned and unburned plots. *Aphaenogaster carolinensis*, the most abundant species, was collected approximately equally between burned and unburned plots. However, some “common” species (those represented by > 1 occurrence) were collected exclusively from one treatment (e.g., *Lasius aphidicola* was collected only in unburned plots, while *Formica subcericea* was collected only in burned plots).

The sampling year had a significant effect on ant species richness (p-value = < 0.001; Table 1; Fig. 5.), with the first year of sampling yielding a greater number of unique occurrences than a second year of sampling of a plot. However, treatment did not have a significant effect on unique occurrences (p = 0.46; Fig. 6) and there was no treatment x year interaction (p = 0.27; Table 1).

Ant communities in burned plots were significantly different from communities in unburned plots (F = 23.94, df = 1, P = 0.001 and F = 2.25, df = 1, P = 0.001, respectively; Table 2; Fig. 3). Sampling year also resulted in a distinction between communities that were sampled during the first year of collection and communities sampled a year following the initial burn (Table 2; Fig. 3). Year and treatment had no significant effect in the PERMDISP test (F = 0.22, P = 0.66; F = 0.12, P = 0.72; Table 3), indicating that observed differences were attributable to the main effects, not dispersion within the data.

Ant community composition in the burned plot was significantly different from control site in 2021, but there was no treatment effect on composition in 2022 (Table 4). Ant composition also differed between the two burn plots (Table 4). Ant composition also differed between the 2021 and 2022 samples from a site that was burned in 2021, and between its associated control samples (Table 4).

Discussion

The result of this study generally agrees with those of similar studies, specifically, that prescribed fire alters epigeic ant communities (Izhaki 2003, Underwood and Quinn 2010). However, it contrasts with other studies indicating that ant species composition does not differ between burned and unburned plots (Verble and Yanoviak 2013). Pitfall traps measure ant activity, so these results can also be explained by a decrease in ant activity, not necessarily extermination of local species.

I suspect that the differences in year are due to differences in burn intensity between 2021 and 2022 (Table 4). The 2022 burn appeared to be of a much lower intensity. Specifically, the vegetation appeared less burnt and there was more leaf litter present following the 2022 burn in vs. the 2021 burn. However, these differences in anecdotal and speculative at this point because supporting data regarding fire intensity are lacking. Previous studies determined that prescribed burn intensity influences ant ecology (Underwood and Quinn 2010, Verble 2012).

The overall negative effect of prescribed burning on ant species richness has been illustrated in similar systems (Izhaki et al. 2003, Underwood and Quinn 2010). Contrasting studies, however, indicate that prescribed burning can increase ant occurrences and species richness (Beaumont et al. 2012, Banschbach and Ogilvy 2014) or have temporary effects on ant species composition (Verble and Yanoviak 2013). The variety of responses of epigeic ants to prescribed fire might reflect ecosystem-level differences (Vasconcelos et al. 2016). The results of this study suggest that most ant species are resilient to prescribed burns. However, pitfall traps do not provide a complete picture on the effects of fire on ants, thus further study of individual species responses is needed to clarify the short- and long-term ecological effects of fire.

The differences between communities at the 2021 burned and unburned plots one year after the initial burn suggest that ant assemblages require more than a year to recover from fire. Communities within the 2021 burn plot are not resilient enough to rebound within a year. This finding contrasts with a similar study that suggests epigeic ant communities in deciduous forest are relatively resilient and can return to normal activity levels less than a year after the initial prescribed burn (Verble and Yanoviak 2013). The significant difference in the 2021 control plot from 2021 to 2022 could be due to a variety of factors, including source/sink dynamics from the adjacent burn plots, natural turnover, or environmental differences between the years (Crist 2009).

In summary, fire intensity likely influences ant activity levels, but future measurements of the intensity of prescribed burns is necessary to confirm this. As prescribed burning is increasingly being added to forestry management to promote plant diversity (Richter et al. 1982, Mitchell et al. 2006), it is important to document its effects on common consumer taxa like ants. Although the effect of prescribed burning on ants has been studied in many other ecosystems (Izhaki et al. 2003, Andersen et al. 2009, Underwood and Christian 2009, Houdeshell et al. 2011, Beaumont et al. 2012, Menke et al. 2015, Bonoan and McCarthy 2022), additional data are needed for temperate deciduous forests, as they are expansive but steadily declining ecosystems (Loucks 1998).

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Tables

Table 1. Output of generalized linear mixed effect model with a negative binomial distribution.

Effect	Term	Estimate	Std. Error	Z-value	p-value
Intercept	Fixed	2.21	0.19	11.36	**<0.001**
Year	Fixed	-0.52	0.15	-3.52	**<0.001**
Status	Fixed	0.20	0.27	0.75	0.46
Year:Status	Interaction	-0.23	0.21	-1.11	0.27
Site	Random	0.04	NA	NA	NA

Table 2. Permutational multivariate analysis of variance (PERMANOVA) output.

Effect	Term	df	Sums Squares	F-value	P(perm)
Year	Fixed	1	2.83	23.94	**0.001**
Status	Fixed	1	0.27	2.25	**0.001**
Year:Status	Interaction	1	0.14	1.16	0.25
Site	Random	NA	NA	NA	NA
Residuals	NA	95	11.22	NA	NA
Total	NA	98	14.45	NA	NA

Table 3. Permutational multivariate analysis of dispersion (PERMDISP) output.

Effect	Term	Sums Squares	F-value	P(perm)
Year	Fixed	0.005	0.22	0.66
Residuals	NA	2.04	NA	NA
Status	Fixed	0.002	0.12	0.72
Residuals	NA	0.02	NA	NA

Table 4. Output of post-hoc test with a Bonferroni correction comparing burned and unburned communities with the 2021 and 2022 prescribed burns. B1 refers to the 2021 burn and B2 refers to the 2022 burn. Yr1 refers to the first sampling period and Yr2 refers to the second sampling period. B refers to burned forest and U refers to unburned forest.

Pairs	df	Sums Squares	F-value	R^2	p-value	adjusted p-value
B1_Yr1_B vs. B1_Yr2_B	1	2.36	34.05	0.47	0.001	0.005
B1_Yr1_B vs. B2_Yr1_B	1	1.87	27.41	0.42	0.001	0.005
B1_Yr1_B vs. B1_Yr1_U	1	0.31	5.58	0.09	0.001	0.005
B2_Yr1_B vs. B2_Yr1_U	1	0.25	2.58	0.13	0.034	0.17
B1_Yr1_U vs. B1_Yr2_U	1	2.25	33.84	0.47	0.001	0.005

Table 5. Summary of taxa collected and their corresponding occurrences in pitfall traps in burned or unburned forest.

Taxon	Burned	Unburned
Amblyoponinae		
Amblyoponini		
Stigmatomma pallipes	4	3
Formicinae		
Camponotini		
Camponotus americanus	129	54
Camponotus pennsylvanicus	78	127
Camponotus chromaiodes	124	81
Camponotus subbarbatus	8	16
Camponotus castaneus	1	0
Formicini		
Formica pallidefulva	1	0

Formica subsericea	33	2
Formica subintegra	1	0
Lasiini		
Lasius aphidicola	0	19
Prenolepis imparis	2	0
Plagiolepidini		
Brachymyrmex depilis	2	7
Myrmicinae		
Attini		
Strumigenys ohioensis	0	1
Strumigenys talpa	1	1
Crematogastrini		
Crematogaster cerasi	60	52
Crematogaster pilosa	25	19
Formicoxenini		
Temnothorax curvispinosus	1	0
Temnothorax schaumii	0	1
Stenammini		
Aphaenogaster carolinensis	497	487
Aphaenogaster fulva	24	43
Ponerinae		

Ponerini		
Ponera pennsylvanica	7	6
Proceratiinae		
Proceratiinae		
Proceratium chickasaw	1	0

Figures

Fig. 1. Pitfall trap placed in burned portion of forest.

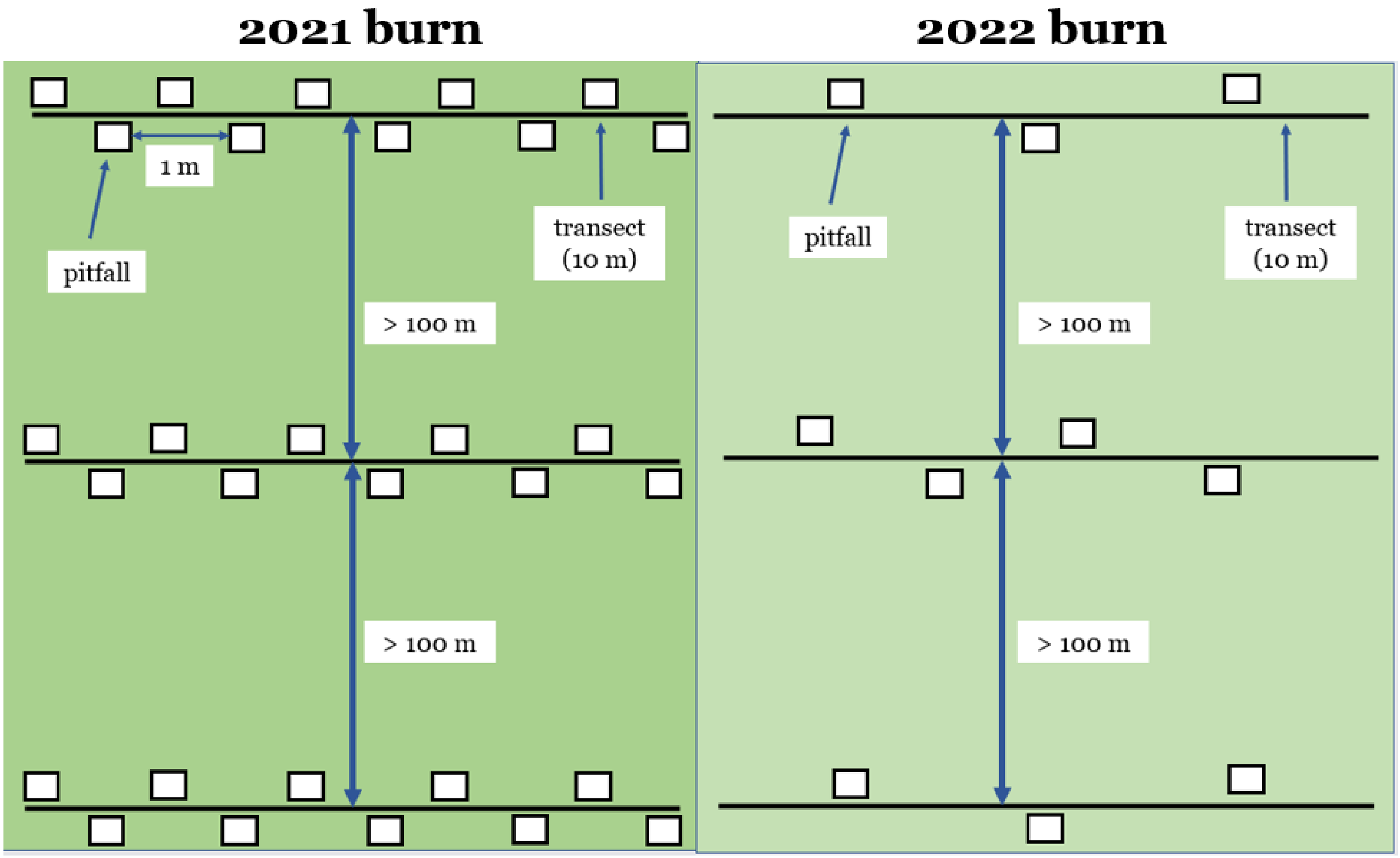

Fig. 2. Pitfall setup for both forest treatments (burned vs. unburned). The first panel symbolizes the layout of pitfall traps during sampling of the 2021 burn among burned and unburned plots. The second panel symbolizes the layout during 2022 sampling of the burned and unburned 2022 burn plots, and continued sampling of burned and unburned 2021 burn plots. White rectangles represent individual pitfall traps and black lines represent 10 m transects.

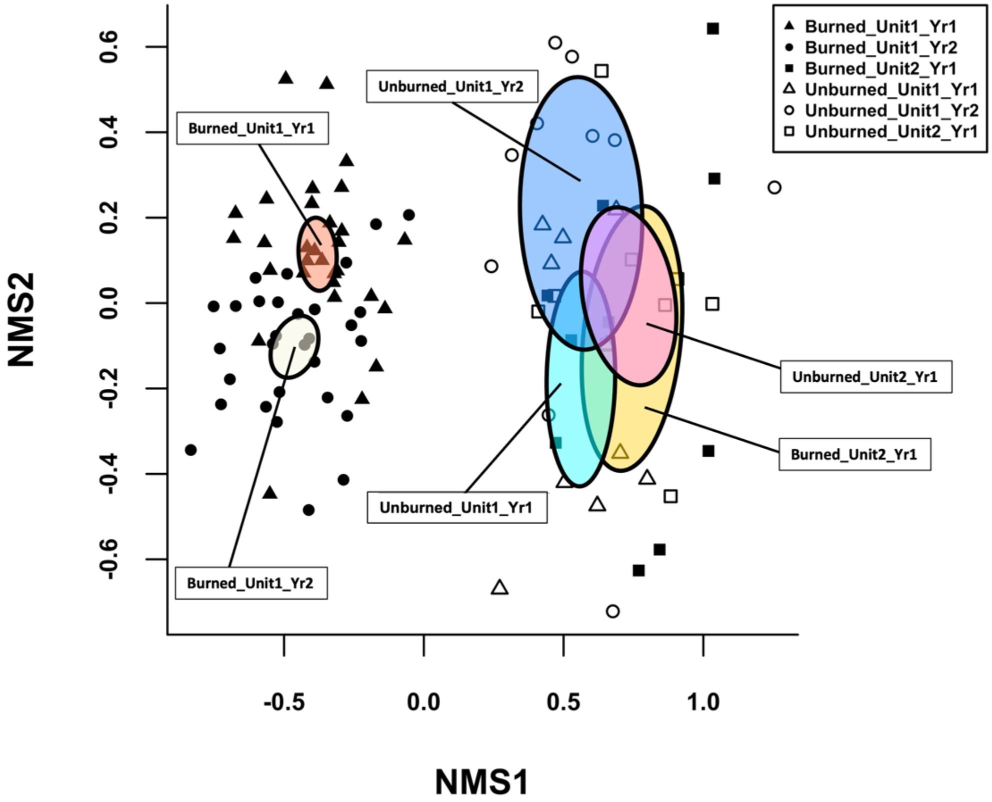

Fig. 3. Non-metric multidimensional scaling (NMDS) ordination plot illustrating differences in ant species composition between forest treatment (burned vs. unburned) and collection year (2021 vs. 2022).

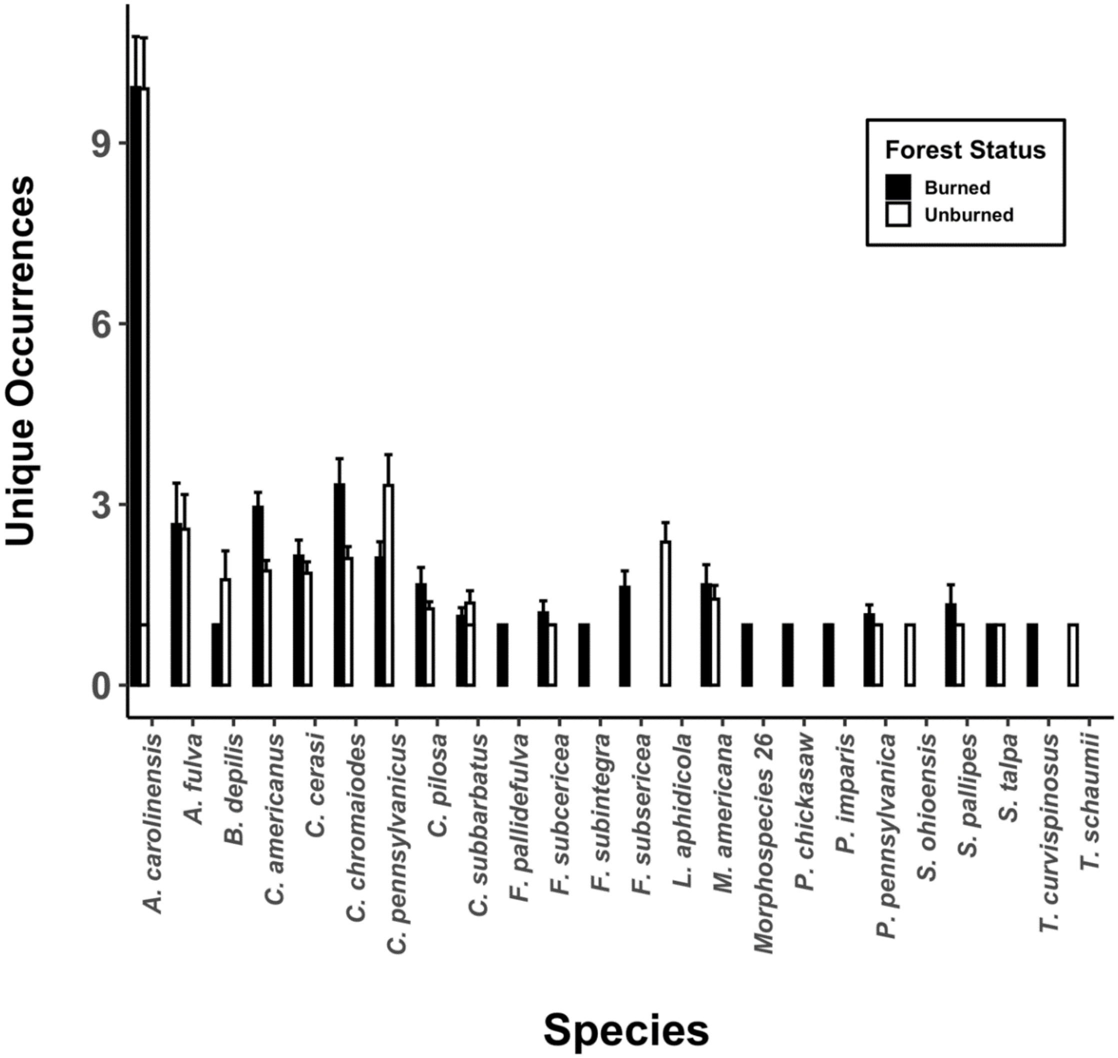

Fig. 4. The number of unique species occurrences (± 1 SE error bars) by each ant species collected in this study.

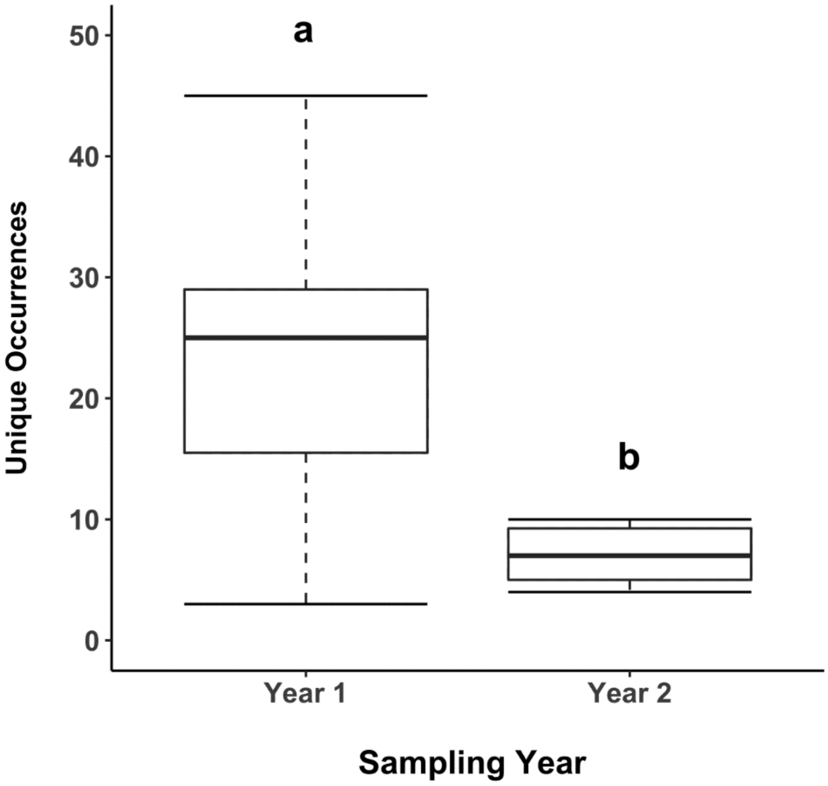

Fig. 5. Box and whisker plot illustrating the pooled data for each pitfall trap and the unique occurrences across sampling years.

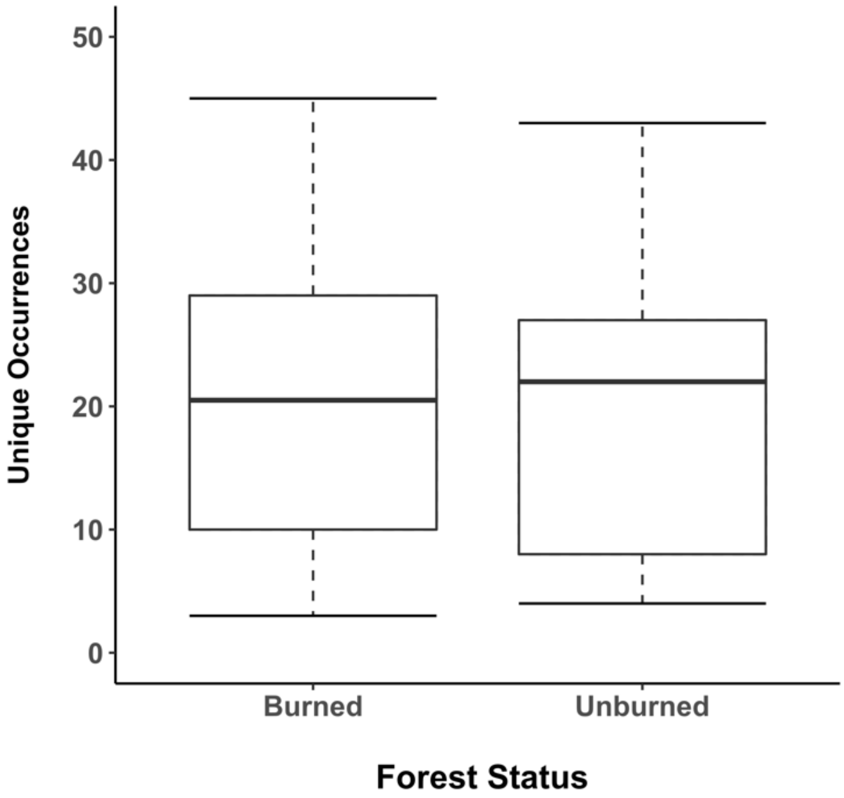

Fig. 6. Box and whisker plot illustrating the pooled data for each pitfall trap and the unique occurrences in burned and unburned forest traps.