**ANANZA MARA RABELLO**

# LAND USE CHANGES AFFECT ANT ASSEMBLAGE AND ECOLOGICAL FUNCTION IN DIFFERENT WAY IN BRAZILIAN SAVANNA

**LAVRAS - MG**
**2017**

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Dra. Carla Rodrigues Ribas
Orientadora

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# LAND USE CHANGES AFFECT ANT ASSEMBLAGE AND ECOLOGICAL FUNCTION IN DIFFERENT WAY IN BRASILIAN SAVANNA

# MUDANÇAS NO USO DA TERRA AFETAM ASSEMBLEIA DE FORMIGAS E FUNÇÃO ECOLÓGICA DE MANEIRA DIFERENTE NA SAVANA BRASILEIRA

Tese apresentada à Universidade Federal de Lavras, como parte das exigências do Programa de Pós-Graduação em Ecologia Aplicada, para a obtenção do título de Doutor.

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**LAVRAS - MG**
**2017**

*Aos meus amados pais, Ronaldo e Silvia, por todo apoio emocional e amor incondicional*

DEDICO

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## RESUMO GERAL

As savanas tropicais são altamente diversas em termos de riqueza de espécies e serviços ecossistêmicos, e ainda assim estão sujeitas a intensa pressão antrópica com altas taxas de perda e degradação de habitat. No Cerrado (savana brasileira) mais da metade das áreas nativas tem sido transformadas em pasto e agricultura, e a maioria dos estudos têm avaliado a estruturação das comunidades biológicas e o impacto das mudanças do uso da terra em apenas um tipo de fitofisionomia e focando somente no processo de desmatamento. Tal fato dificulta avanços em compreender alterações nas comunidades biológicas da fauna nativa do Cerrado em geral. Assim, essa tese objetivou avaliar fatores estruturando a comunidade biológica em escala local e de paisagem, e os impactos das mudanças no uso da terra (Eucalipto e pasto) sobre a comunidade e função ecológica de formigas em três fitofisionomias do Cerrado (campo limpo, cerrado sensu stricto e cerradão) através dos processos de perda de árvores e arborização (ganho de árvore). Eu coletei as formigas em pitfall epigeico e formigas removendo sementes (função ecológica) em áreas nativas de Cerrado, Eucalipto e pasto em cada fitofisionomia. No capítulo 1 eu averiguei os fatores que influenciam a assembleia de formigas em habitat nativos em diferentes fitofisionomias do Cerrado em escala local e de paisagem. Obtive que a assembleia de formigas em habitats nativos é predominantemente influenciada por fatores de paisagem, indicando a importância do entorno das áreas nativas e de incluir fatores de paisagem em estratégias de manejo e conservação. No capítulo 2 eu utilizei as abordagens taxômica e de guildas para avaliar se as mudanças no uso da terra, por perda de árvores e arborização, afetam as espécies e guildas de formigas de maneira similar. Encontrei que a resposta da estrutura taxonômica ás mudanças do uso da terra não necessariamente leva a mesma resposta da estrutura de guildas independente do tipo de mudança. Isso mostra que ambas abordagens, taxonômica e guildas, contribuem para melhorar nossa compreensão sobre ecologia de comunidades na conversão de habitats nativos em agroecossistemas. No capítulo 3 eu examinei os impactos das mudanças no uso da terra na remoção de sementes pelas formigas e se esses impactos estavam correlacionados com mudanças nos atributos do habitat. Observei que os impactos na remoção de sementes depende do tipo de mudança no uso da terra (arborização ou perda de árvores) e estão correlacionados com a similaridade dos atributos do habitat entre agroecossistemas e habitat nativo. De forma geral, essa tese mostra que a assembleia de formigas é dependente de fatores de paisagem e que as mudanças no uso da terra, avaliadas simultaneamente em diferentes fitofisionomias, apresentam diferentes impactos na assembleia de formigas e na sua função ecológica de remoção de sementes. Portanto, identificar os fatores estruturando a assembleia de formigas, e em que escala espacial os mesmos atuam, e os impactos das mudanças do uso da terra em diferentes fitofisionomias pode contribuir para aprimorar nosso conhecimento sobre as respostas das comunidades biológicas e estratégias de manejo aplicado e conservação no Cerrado.



**Palavras-chave:** Cerrado. Escalas espaciais. Agroecossistemas. Guildas. Remoção de sementes. Atributos do habitat.

## GENERAL ABSTRACT

Tropical savannas are highly diverse in terms of species richness and ecosystem services, yet are subject to intense anthropogenic pressure with high rates of habitat loss and degradation. In the Cerrado (Brazilian savanna) more than half of native areas have been transformed into pasture and agriculture, and most studies have evaluated biological communities structuring and impacts of land use changes within one vegetation type and focusing only on deforestation process. Such fact raises difficulties advances in understanding alterations in biological communities of native fauna from Cerrado in general. Thus, this thesis aimed to evaluate factors structuring biological community in local and landscape scales, and the impacts of land use changes (*Eucalyptus* plantation and planted pasture) on ant community and ecological function in three Cerrado vegetation types (grassland, savanna and savanna-forest) through both tree less and afforestation (gain of tree) processes. I sampled ants using epigaeic pitfall and ants removing seeds (ecological function) in areas of native Cerrado, *Eucalyptus* and planted pasture in each vegetation type. In chapter 1 I assessed the factors influencing ant assemblage in native habitats in different Cerrado vegetation types at local and landscape scales. I observed that ant assemblage in native habitats is predominantly influenced by landscape factors, indicating the importance of surrounding native habitats and to include landscape factors in management and conservation strategies. In chapter 2 I used taxonomic and guild approaches to evaluate if land use changes, by tree loss and afforestation, affect ant species and guilds in a similar way. I found that response of taxonomic structure to land use changes do not necessarily lead to the same response of guild structure regardless of the type of change. This shows that both approaches, taxonomic and guild, contribute to improve our understanding about community ecology in conversion of native habitats into agroecosystems. In the chapter 3 I examined the impacts of land use changes on seed removal by ants and if these impacts were correlated with changes in habitat attributes. I observed that these impacts on seed removal depend on the type of land use change (afforestation or tree loss), and are correlated to the similarity in habitat attributes between agroecosystems and native habitat. Overall, this thesis shows that ant assemblage is dependent on landscape factors and that land use change, when evaluated simultaneously across different vegetation types, have different impacts on ant assemblage and their ecological function of seed removal. Therefore, identifying factors structuring ant assemblage, as well as the spatial scale they operate, and the impacts of land use change in different vegetation types may contribute to improve our understanding about the response of ecological communities and applied management and conservation strategies in the Cerrado.



**Key-words:** Cerrado. Spatial scales. Agroecosystems. Guilds. Seed removal. Habitat attributes.

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**SUMÁRIO**

# PRIMEIRA PARTE

# 1 INTRODUÇÃO GERAL

Um dos maiores desafios na Ecologia de Comunidades atualmente é prever como a alteração do habitat afeta a biodiversidade. Isso se torna ainda mais importante e desafiador em regiões tropicais onde a alteração do habitat devido à mudanças no uso da terra tem levado a perda de biodiversidade e extinção de espécies, e consequentemente ao declínio de muitas funções ecológicas e serviços ecossistêmicos (MELO et al., 2013; NEWBOLD et. al., 2015). As regiões tropicais apresentam a dicotomia em conter a maior biodiversidade do mundo e, ao mesmo tempo, as maiores taxas de mudanças no uso da terra (GARDNER, 2000; ROMDAL et al., 2013). As florestas nativas tropicais já perderam praticamente 70% da sua cobertura florestal original devido às mudanças no uso do solo (PERFECTO; VANDERMEER, 2008; MELO et al., 2013). Essas taxas continuam crescendo e gerando severas mudanças na configuração das paisagens tropicais, que se tornam altamente fragmentadas e vulneráveis à distúrbios naturais e antrópicos (GARDNER et al., 2009; BARLOW et al., 2016).

Em regiões tropicais, a expansão de práticas de agricultura intensiva, silvicultura, pastagem para gado, corte seletivo e mineração são as principais mudanças no uso da terra responsáveis pela perda e fragmentação de habitat e alteração das comunidades e interações ecológicas (SANO et al., 2010; HADDAD et al., 2015). No Brasil, as mudanças no uso da terra mais comuns são: pastagem para gado, agricultura intensiva e silvicultura (ex. plantação de *Eucalyptus*), em ordem de extensão (IBGE 2012). As alterações de habitat causadas por mudanças no uso da terra no Brasil têm sido mais avaliadas na Amazônia e Mata Atlântica por serem os dois maiores biomas florestais brasilieros com alta taxa de aletração de habitat, e menos atenção tem sido dada a outros importantes biomas como o Cerrado (KLINK; MACHADO, 2005; METZGER, 2009; SLOAN et al., 2014; SOLAR et al., 2015).

O Cerrado é o segundo maior bioma do Brasil em território ocupado, e é altamente heterogêneo em termos de biodiversidade e fitofisionomias (SILVA; BATES, 2002; KLINK; MACHADO, 2005). Podemos constatar essa heterogeneidade fitofisionômica pela presença de vegetações nativas que vão de campos limpos à florestas (OLIVEIRA-FILHO; RATTER, 2002; SANO et al., 2010). Apesar disso, nas últimas décadas, o Cerrado têm sofrido severas mudanças no uso da terra com a introdução de pastagem para gado e agricultura mecanizada que levaram a perda de mais da metade da sua cobertura original (SANO et al., 2010; GRECCHI et al., 2014). Essas mudanças no uso da terra, a uma velocidade rápida, geraram danos ambientais alarmantes como fragmentação, perda de biodiversidade, espécies invasoras e mudanças no regime de fogo (KLINK; MACHADO, 2005; PACHECO et al., 2013). Por

isso, atualmente, o Cerrado é considerado um dos *hotspots* mundiais e a savana tropical mais ameaçada do mundo (MYERS, 2000; SILVA; BATES, 2002).

Em geral, os agroecossistemas apresentam como efeito típico da mudança do uso da terra a alteração dos atributos do habitat (ex. estrutura de dossel, cobertura de gramíneas) que possuem considerável influencia na distribuição, diversidade e interações das espécies (LASSAU; HOCHULI, 2004; TEWS et al., 2004). No Cerrado, devido a heterogeneidade de fitofisionomias, as alterações nos atributos do habitat podem acontecer por dois processos: desmatamento, com a remoção de árvores em fitofisonomias que são originalmente florestais; e arborização, com o ganho de árvores em fitofisionomias que são originalmente campos ou savanas (BREMER; FARLEY, 2010; VELDMAN, 2015). Ambos processos provocam mudanças na disponibilidade de recursos, condições ambientais e produtividade, afetando a dinâmica das comunidades (NORFOLK et al., 2012, DE LA MORA et al, 2013).

Entretanto, os agroecossistemas que contêm maior similaridade nos atributos do habitat com as áreas nativas que eles circundam apresentam menor impacto sobre a biodiversidade local (FRIZZO; VASCONCELOS, 2013, LIVINGSTON et al., 2013). Tal predição exige avaliação e definição dos atributos do habitat que definem e compõem os habitats nativos e modificados, e também que as particularidades nos atributos de habitats nativos de diferentes fitosionomias sejam reconhecidas (OVERBECK et al., 2007; VELDMAN, 2016). Nesse segundo ponto, o Cerrado têm se mostrado em desvantagem, pois ainda existe uma falta de conhecimento, por parte de alguns pesquisadores e organizações ambientais, das características e particularidades de cada fitofisionomia, gerando percepções equivocadas sobre sua ecologia, valor de conservação e localização (KLINK; MACHADO 2005; PACHECO et al., 2012). Os campos limpos, por exemplo, são muitas vezes considerados como oportunidades de restauração florestal ou não são devidamente diferenciados de pastos; e as florestas do Cerrado não são muitas vezes consideradas como parte do bioma Cerrado (PARR et al., 2014).

Além dos atributos do habitat em escala local, que influenciam a diversidade e função ecológica das espécies, os fatores em uma escala de paisagem (ex. quantidade de habitat nativo, de habitats transformados) também apresentam influencia sobre a biodiversidade e conservação de espécies em habitats nativos em paisagens modificadas por agroecossistemas (DE LA MORA et al., 2013; GOLLAN et al., 2014; SOGA et al., 2015). Algumas espécies nativas são capazes de persistir em paisagens modificadas que apresentem fatores favoráveis a sua imigração entre habitats nativos, diminuindo os riscos de que extinções locais se tornem regionais (MITCHELL et al., 2015). Por isso, é importante considerar, além dos fatores

locais, os fatores que atuam em escala de paisagem para gerar avanços no conhecimento sobre as consequências da mudança no uso da terra sobre a biodiversidade, e sobre alternativas de manejo dos agroecossistemas com o intuito de diminuir seus impactos (DAUBER et al., 2005; DE LA MORA et al., 2013).

Em vista as altas taxas de conversão de áreas nativas de Cerrado em agroecossistemas e a atual escassez de estudos abordando como essas conversões afetam a biodiversidade das diferentes fitofisionomias do Cerrado, abordaremos aqui a avaliação da estrutura da comunidade e função ecológica usando as formigas como modelo de estudo. Assim, esta tese apresenta três capítulos originais organizados como manuscritos a serem enviados para revistas científicas indexadas, e por isso, estão escritos em língua inglesa. Em conjunto, os capítulos abordam os importantes temas acima apresentados envolvendo as mudanças no uso da terra em diferentes fitofisionomias do Cerrado. Além desses capítulos, a tese conta com a presente introdução geral e conclusão geral.

No primeiro capítulo nós investigamos se a assembleia de formigas em habitats nativos é mais influenciada por fatores locais (intrínsecos do habitat nativo) ou por fatores de paisagem nas diferentes fitofisonomias do Cerrado. No segundo capítulo verificamos o padrão de resposta de duas abordagens, taxonômica e guildas, frente a conversão de áreas nativas de Cerrado em agroecossistemas (pasto e plantação de *Eucalyptus*) em três diferentes fitofisionomias, através dos processos de perda de árvores e arborização. No terceiro capítulo nós buscamos responder se o grau do impacto do pasto e plantação de *Eucalyptus* na função ecológica de remoção de sementes depende do tipo de fitofisionomia estudada e se está correlacionado com os atributos do habitat.

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# SEGUNDA PARTE - ARTIGOS

# ARTIGO 1

## Ant assemblage in different Cerrado vegetation types: assessing local and landscape drivers

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## ABSTRACT

Understanding the factors structuring biodiversity in native habitats with agriculture matrix is necessary to consider the different spatial scales, local and landscape, at such factors operate for conservation purposes. In this context, most of information comes from studies in forests systems, and very little is known about the factors influencing species diversity in savannas within agriculture landscape. We sought to start fill this gap by identifying some factors, local and landscape, structuring ant assemblage in native habitats in three Cerrado vegetation types (grassland, savanna and savanna-forest). We used five areas of native habitats in each vegetation type for sampling local and landscape factors. We found that local factors have no influence on ant richness and composition in native habitats regardless of the vegetation type. Landscape factors, such as native habitat cover, planted pasture cover and connectivity among native habitats, influenced ant diversity in grassland, savanna and savanna-forest respectively. Such findings show that ant assemblage in native habitats is predominantly influenced by anthropogenic matrices surrounding native habitats in all Cerrado vegetation types. Our data highlight the importance of preserving native habitats patches and permeable matrices for biodiversity conservation. Local and landscape factors have different importance in structuring species diversity between forest and savanna systems in tropical region, indicating that such factors should be carefully evaluated in different biomes. This study suggests that environmental and farm managers should therefore consider landscape factors when seeking to enhance and conserve biodiversity in tropical savannas within landscape highly threatened by land use changes.


**Key-words:** Cerrado, multiscale, connectivity, Formicidae

## INTRODUCTION

Land use change intensification is the main cause of biodiversity loss and ecosystem functioning alteration across the world (Hooper et al. 2005; Newbold et al. 2015). Particularly, the tropical regions remain at the forefront of high levels of land use change by deforestation, fragmentation, agriculture intensification, among other anthropogenic impacts (Gardner et al. 2009; Melo et al. 2013). Meanwhile, native habitats are in decline and increasingly fragmented (FAO 2010).

Intensive agricultural and agroforestry practices have resulted in modification and destruction of native habitats and consequently in landscape alteration (Vandermeer and Perfecto 2007). Habitat modification can alter native habitat quality which is driver of community richness and composition and an environmental filter determining local communities (Hill et al. 2008; Brudvig 2011). Landscape alteration decreases native habitat amount and changes species foraging and dispersal success, then, may cause negative effects on biodiversity in native habitats (Fahrig 2003). Therefore, to evaluate the effects of land use change on native habitats and their native communities it is necessary to consider the different spatial scales at which such effects operate: both local and landscape scales.

Local factors have substantial influence on biological communities due to alteration in local habitat attributes such as soil moisture, tree density and canopy cover (Dauber et al. 2003; Frizzo et al. 2013; Marín et al. 2016). In addition, an increasing number of studies have found the influence of landscape factors on biodiversity in several regions, including the tropics, dominated by agroecosystems (e.g. Batáry et al. 2010; Gollan et al. 2014). However, the importance of local and landscape factors on biodiversity conservation is still little examined simultaneously. In this context, most studies have been done in temperate zones (e.g. Dauber et al. 2005; Schmidt et al. 2008; Woltz et al. 2012; Soga et al. 2015) and, in tropical regions, such studies have focused only on forest vegetation type (Barlow et al. 2016; Marín et al. 2016; Solar et al. 2016). Assessing the relative importance of local and landscape factors on biodiversity in native habitats may help to establish alternative management actions for the conservation of biodiversity in landscapes threatened by agroecosystems (Dauber et al. 2005).

In this sense, the Cerrado (brazilian savanna) is an ideal system to study the factors influencing biodiversity at both local and landscape scales in agricultural landscapes. The Cerrado has different native vegetation types, varying from grasslands to savannah-forests, with different local habitat attributes (i.e. canopy cover and grass cover) and all of them under

high rates of land use change by agroecosystems (Grecchi et al. 2014). Such local habitat attributes differences allow comparing the importance of local factors affecting species in different vegetation types. As around half of the Cerrado has been replaced by agroecosystems leading to high level of landscape degradation and biodiversity loss, the Cerrado is appropriate to evaluate native habitat alteration and degradation by land use change at landscape scale (Melo et al. 2013). Furthermore, very little is known about the importance of landscape factors affecting biodiversity in Cerrado´s agricultural landscapes (Pacheco et al. 2012). Knowing about the factors affecting biodiversity in both local and landscape scales is very important for the Cerrado which is affected in terms of habitat quality and landscape structure by agroecosystems.

To evaluate and compare which factors influence biodiversity in native habitats within agricultural landscape, at local and landscape scales, ants are excellent model organisms since they are abundant, diverse, high sensitive to habitat changes and respond rapidly to human disturbances (Folgarait 1998; Andersen and Majer 2004; Majer et al. 2007; Philpott et al. 2010). Yet despite ant importance, ant richness and composition respond to both local (i.e. litter biomass) and landscape (i.e. distance from forest, amount of forest) factors in agricultural landscapes (De la Mora et al. 2013; Solar et al. 2016). However, most studies have addressed ant response evaluating local and landscape factors only in forest native vegetation types. Thus, we still lack view about factors regulating ant assemblage in different savanna vegetation types simultaneously to best understand drivers promoting ant persistence and to provide a basis for predicting the biodiversity consequences in tropical landscapes highly threatened by agrosystems.

As native habitats are core of biodiversity conservation strategies and are highly susceptible to agrosystems impacts, it is important to understand the factors regulating species diversity in native habitats within an agricultural landscape. Here we focus on understanding the factors, local and landscape, regulating ant assemblages in native habitats in different Cerrado vegetation types (grassland, savannah and savannah-forest). We expected that local factors are more important than landscape in regulating the ant diversity in native habitats in all Cerrado vegetation types, since ants are strongly affected by local microclimatic conditions (Lassau and Hochuli 2004).

## MATERIAL AND METHODS

*Site description*

Our study region was located in three cities of Minas Gerais state, Brazil, in the Cerrado biome: Itutinga (21°25′39.9″S 44°34′27.4″W), Itumirim (21°13′55.7″S 44°48′39.3″W) and Boa Esperança (21°04′16.8″S 45°36′36.6″W). The climate in the region is characterized as Equatorial savanna with dry winters (April to September) and wet summers (October to March). Annual mean temperature and precipitation are 20ºC and 1500mm respectively. Sampling took place in the rainy season from January to March 2014. The landscape in the region is composed of heterogeneous mosaics of native habitat, human settlement, pasture, *Eucalyptus* plantation and water (Melloni et al. 2013).

We carried out this study in native habitats on private farms in three Cerrado vegetation types: grassland (*campo limpo*) with no woody species, savanna (*cerrado sensu stricto*) with grass, closed shrub and scattered trees, and savanna-forest (*cerradão)* with tall trees and canopy cover developed (Silva and Bates 2002; Oliveira-Filho and Ratter 2002).

*Local factors*

In each Cerrado vegetation type, we selected five sites of native habitat where we installed ten sampling points, 20 m apart and 50 m from edge, and then we measured in each sampling point the following local habitat attributes: canopy cover, litter diversity and herbaceous ground cover. Queiroz et al. no prelo observed these local habitat attributes affecting ant diversity in the same study site. We estimated the percentage of canopy cover using a fish-eye lens attached to a camera positioned at height 1.5 m; digital images were analyzed in a software gap light analyzer (GLA) 2.0 to obtain openness percentage (Frazer et al. 1999). We collected litter using a 25 x 25 cm quadrat and, afterwards, we counted the number of different categories of leaves, branches and sticks in order to calculate the litter diversity with Inverse Simpson index (modified from Queiroz et al. 2013). Finally, we estimated the percentage of herbaceous ground cover using a 1 x 1 m quadrat at each sampling point.

*Landscape factors*

In all Cerrado vegetation types, we used satellite images from basemap imagery (2008 to 2011) with reference scale of 1:5.000 and 5 m resolution to classify and characterize land uses. We used 300 m buffer surrounding each native habitat, from the transect, to measure

landscape variables (Fig. A1.2). We used such buffer based on ant foraging distances (Traniello 1989).

In each vegetation type, by using V-LATE and Patch Grid software, we used only native habitats correspondent for vegetation type sampled and we measured the amount of native habitat cover (NHC), agriculture cover (AC), planted pasture cover (PC) and mean proximity index of native patches (MPI) within each landscape in each vegetation type separately as our landscape variables. The mean proximity index is a metric of connectivity that is calculated by summing the area of all native patches within 300 m buffer, divided by the square distances of all native patches within the same buffer for landscape value (Gustafson and Parker 1992). Proximity index of native patches and native habitat cover have been documented to influence ant assemblage (De la Mora et al. 2013; Solar et al. 2016). Considering that matrixes surrounding native habitats may influence biodiversity (Escobar et al. 2008), we also chose agriculture and planted pasture cover as landscape factors for being the main matrixes in the three Cerrado vegetation types.

*Ant diversity sampling*

In each Cerrado vegetation type we sampled ants in five sites of native vegetation at the same ten sampling points where we measured the local factors. In each sampling point we installed one unbaited epigaeic pitfall trap (8 cm in diameter and 20cm in deep). Each pitfall trap contained 200 ml solution of water (99%), liquid soap (0.6%) and salt (0.4%) and was covered to protect against sun and rain; all pitfall traps were kept opened for 48h.

We first sorted the ant specimens to genus following Baccaro et al. (2015) and then to morphospecies. Ant morphospecies were checked and where possible identified as described species by T.S.R. Silva and G. Camacho (Laboratório de Sistemática e Biologia de Formigas, Universidade Federal do Paraná, Brazil) following De Andrade and Baroni-Urbani (1999), Lattke et al. (2007), Mayhé-Nunes and Brandão (2002 and 2005), and Wilson (2003). Voucher ant specimens are held at the Laboratório de Ecologia de Formigas, Universidade Federal de Lavras, Brazil.

*Data analysis*

First, all local and landscape variables were standardised to zero mean and unit standard deviation prior to all analysis to avoid biased results and allow us to compare variables with different scales. Second, for each vegetation type, we examined if there was some correlation

and avoid collinearity among local and landscape variables (Pearson $\rho > 0.70$) using Pearson correlation coefficient. In grassland, proximity index was negatively correlated to ground herbaceous cover (Pearson $\rho = -0.73$), canopy cover correlated to agriculture cover (Pearson $\rho = 0.73$) and pasture cover negatively correlated to native habitat cover (Pearson $\rho = -0.92$) and then proximity index, canopy cover and pasture cover were removed from analyses. In savanna, canopy cover was negatively correlated to ground herbaceous ground cover (Pearson $\rho = -0.89$) and native habitat cover correlated to litter diversity (Pearson $\rho = 0.88$), so canopy cover and native habitat cover were removed from analyses. In savanna-forest, native habitat cover was correlated to canopy cover (Pearson $\rho = 0.93$) and ground herbaceous cover negatively correlated to litter diversity (Pearson $\rho = -0.89$), then native habitat cover and ground herbaceous cover were also removed from analyses.

Then, to evaluate the influence of local and landscape factors on ant richness, we used hierarchical partitioning analysis (Chevan and Sutherland 1991). Hierarchical partitioning is a multiple-regression that tests the independent effects of explanatory variables (local and landscape factors) on response variable (ant richness) through all possible model combinations by changing the order of the explanatory variables. For such analysis, we used *hier.part* package (Walsh and Mac Nally 2007). We conducted Pearson correlation coefficient and Hierarchical partitioning in R version 3.3.0 (R Core Team 2016).

We investigated the influence of each local and landscape factors on ant composition using a distance-based linear model (DistLM) with Jaccard index with presence/absence data. We run DistLM in Primer version 6.0 PERMANOVA+ add-on (Clarke and Gorley 2006).

## RESULTS

We collected a total of 182 ant species within 46 genera and seven subfamilies. The richest subfamily was Myrmicinae with 24 genera and 122 species The richest genera was *Pheidole* (38), *Camponotus* (27) and *Solenopsis* (14). The complete list of ant species is inserted as appendix in chapter 2 in this thesis.

### *Local factors*

Contrary to our expectations, local factors had no influence on ant richness in native habitats in all vegetation types (Fig1.1 a,b,c). The same was observed for ant composition in all vegetation types (Table 1.1).

*Landscape factors*

In grassland, none of landscape variables influenced ant richness in native habitats (Fig.1.1a), but ant composition was influenced by native habitat cover (Table 1.1). Ant richness was positively influenced by pasture cover (z = 1.96) in savannah (Fig. 1.1b) and proximity index (z = 1.78) in savannah-forest (Fig.1.1c). In savanna and savannah-forest vegetation types, landscape factors did not influence ant composition (Table 1.1).

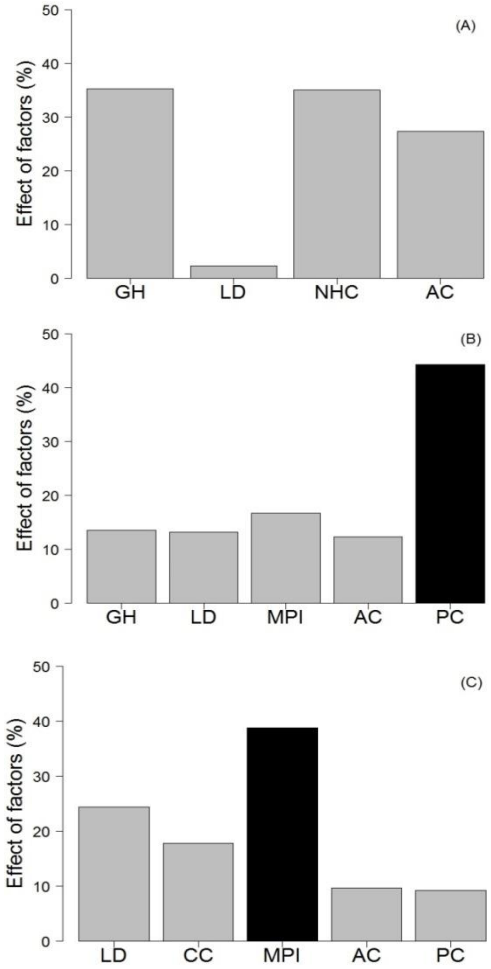


Figure 1.1: Influence of local and landscape factors on ant species richness in native habitats in grassland (A), savanna (B) and savanna-forest (C). Black bars indicate positive and significant influence ($p < 0.05$). GH = herbaceous ground cover, LD = litter diversity, CC = canopy cover, MPI = proximity index, NHC = native habitat cover, AC = agriculture cover, PC = planted pasture cover.

Table 1.1: Influence of local and landscape factors on ant composition in native habitats in different Cerrado vegetation types. Significant influence ($p < 0.05$) is highlighted in bold letters. GH = herbaceous ground cover, LD = litter diversity, CC = canopy cover, MPI = proximity index, NHC = native habitat cover, AC = agriculture cover, PC = planted pasture cover.

| Vegetation type | Explanatory variables | Pseudo F | *p* value | Proportion of explanation |
|---|---|---|---|---|
| Grassland | GH | 1.08 | 0.5 | 0.26 |
| | LD | 0.88 | 0.7 | 0.22 |
| | **NHC** | **1.32** | **0.04** | **0.30** |
| | AC | 0.70 | 0.9 | 0.19 |
| Savanna | GH | 0.98 | 0.6 | 0.25 |
| | LD | 0.86 | 0.8 | 0.22 |
| | MPI | 1.24 | 0.1 | 0.29 |
| | AC | 1.06 | 0.3 | 0.26 |
| | PC | 1.08 | 0.3 | 0.26 |
| Savanna-forest | LD | 0.69 | 0.8 | 0.19 |
| | CC | 1.22 | 0.3 | 0.28 |
| | MPI | 0.91 | 0.6 | 0.23 |
| | AC | 1.38 | 0.1 | 0.31 |
| | PC | 0.54 | 0.9 | 0.15 |

## DISCUSSION

This study provides valuable insights into environmental factors influencing ant assemblage in native habitats, evaluating the importance of local and landscape factors simultaneously in structuring ant assemblage across different Cerrado vegetation types. Our results indicate that ant assemblage in native habitats is predominantly influenced by landscape factors in all Cerrado vegetation types.

Surprisingly, local factors had no influence on ant richness and composition in all Cerrado vegetation types. This is contrary to most studies that have evaluated the role of local and landscape factors on species richness and composition, and have found that local factors have strong influence on species diversity (Meyer et al. 2015; Marín et al. 2016). Different of studies cited before our study was conducted in savanna native systems, therefore, we thought

of three explanations for the greater influence of landscape than local factors. First, local factors may be more important in explaining the differences in species diversity between areas with contrasting land uses (e.g. native and matrix) considering habitat attributes (Queiroz et al. no prelo). Second, in our study, probably there is little difference in local variation among native habitats making such factors less accurate to describe the influence of local factors on local ant assemblage. Third, landscape factors are more important than local ones in a complex landscape (Tscharntke et al. 2012), which is our case since our study region presents a high complex landscape with fragments of native vegetation, planted pasture, *Eucalyptus* plantation, coffee, human settlement and water increasing the influence of landscape factors.

Our study shows that ant assemblage in native habitats is more accurately influenced by landscape factors than local factors. Fischer and Lindenmayer (2007) reported that landscape alteration, with changes in structure and composition of matrix environments due to land use changes have great influence on biological communities in native habitats. Our finding is in line with other studies with butterflies (Soga et al. 2015), ants (De la Mora et al. 2013), diplopods (Dauber et al. 2005) and Orthoptera (Sutcliffe et al. 2014) which found that species respond to landscape factors predicted by patch native area, connectivity and landscape heterogeneity.

In grassland, native habitat cover influenced local ant composition indicating that ant species may benefit from amount of native habitat in the surrounding landscape. Ant species seems to depend on a range of native habitats in a landscape under great agroecosystems pressure, as in our study area which native habitat loss and degradation is common. Native habitat cover may offer higher immigration probability in the landscape, regardless of its configuration, influencing species composition in native habitats (Ives et al. 2011; Fahrig 2013; Sutcliffe et al. 2014). The importance of native habitat cover as predictor of ant composition indicates the value of native habitat for biodiversity conservation in tropical grasslands embedded within human-modified landscapes.

Pasture cover was an important landscape factor positively influencing ant richness in savanna. Such finding suggests that ant species are not restricted only to savanna habitats, and that pasture habitats appear to provide connectivity between native habitat patches. Thus, planted pasture seems to be a permeable matrix providing favorable microhabitat conditions and allowing ant movement between native habitat patches in savanna vegetation type. Generally, converted habitats that do not represent barriers to species dispersal may increase species reproductive success and represent a potential pool of species available to colonise native habitats (Hatfield and LeBuhn 2007; Sutcliffe et al. 2014) contributing to the increase

in species richness. Furthermore, planted pastures share with savanna some important habitat attributes for ants (see Thesis chapter 3) which may increase the connectivity among native habitats and planted pastures, reducing local extinctions within native habitats (Frizzo and Vasconcelos 2013).

In savanna-forest, the connectivity (mean proximity index) positively influenced ant richness in native habitats pointing out that well functionally connected savannah-forests patches favor ant richness. Connectivity between native habitats patches has been shown to play important role in facilitating species movements and maintain populations that cannot survive in small patches (Ives et al. 2011). It has been concluded from studies with other taxa that local species richness depends on connectivity and large-spatial arrangement of native habitat patches in the landscape to serve as refuges and sources of recolonization (metapopulations) (Dauber et al. 2005; Soga and Koike 2013; Soga et al. 2015).

The most important finding of this study is that the ant assemblage in native habitats responds to landscape factors regardless of vegetation type, even landscape factor predictors are not the same among different Cerrado vegetation types. Such finding represents an important message to policy makers and land use planners that landscape factors should be inserted in management schemes and conservation strategies. Given the influence of native cover, planted pasture cover and connectivity on ant assemblage, we highlight the importance of conserving native habitats amount enough and permeable matrices for providing refuge habitats and chance of species movement. Finally, since the information about factors structuring species at different spatial scales in tropical regions is limited on forest systems, our study innovates in elucidating the influence of surrounding agriculture matrix in structuring species diversity in savanna and helps to guide prioritization of savanna conservation efforts.

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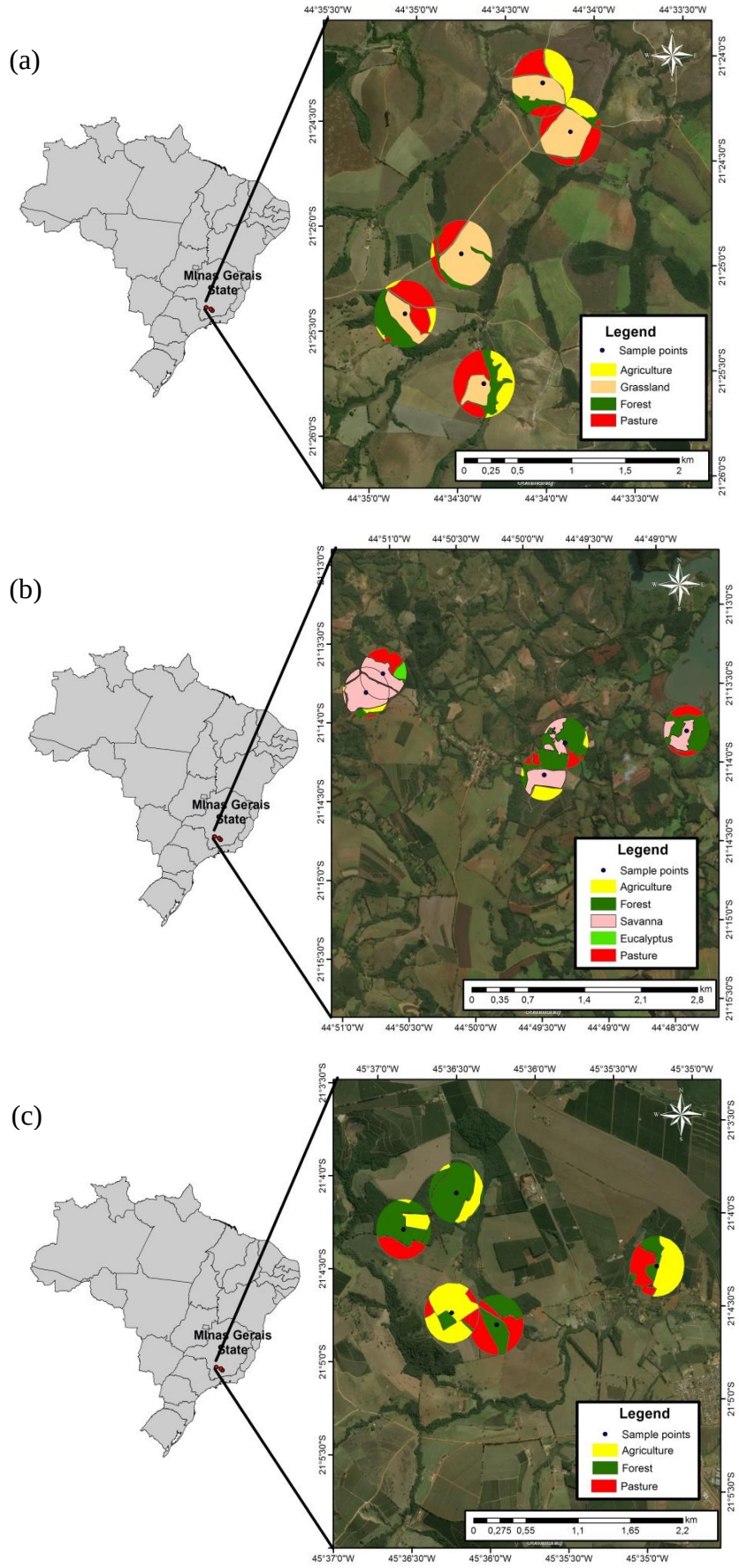


**Figura A1.2:** Buffers around native habitats from the transect within (a) grassland, (b) savanna and (c) savanna-forest vegetation types. Five areas of native habitats were sampled in each vegetation type. Black points represent the transect.

# ARTIGO 2

## Taxonomic and guild approaches reveal differential responses of ant assemblage to land use changes

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**Abstract**

Land use change is well documented to cause the loss of species. However, our understanding of effects of land use change on other aspects of biodiversity, such as guilds, is still limited. We evaluated if land use change (*Eucalyptus* plantation and planted pasture) affects ant assemblage at both taxonomic and guild levels in a similar way across three Cerrado vegetation types (grassland, savanna and savanna-forest). We found that species and guilds respond differently to land use changes regardless of the type of change. We found that land use changes alter only species frequency of occurrence indicating species are more sensitive than guilds to habitat transformation in all Cerrado vegetation types. In grassland, species composition differed between native and converted habitats, but guilds did not differ suggesting a turnover of species within guilds. In savanna and savanna-forest, ant species and guilds had the same response, differing between native and converted habitats, indicating that the loss of species leads to a consequent change in guilds. We observed that, in grassland and savanna, ant species and guild habitat generalists and specialists had different responses to land use change showing higher proportion of habitat specialists in taxonomic than in guild level, and higher proportion of habitat generalists and loss of converted habitat specialists at guild level. Such a finding indicates species differ in their ecology and many guilds are shared among different habitat types. In savanna-forest, species and guild habitat specialists had similar response indicating specific components of ant assemblage structure in native and converted habitats at both taxonomic and guild approaches. Our results demonstrate that the relationship between species and guilds depends on the context (e.g. vegetation type), and they can be independent of each other in response to land use change. We highlight the importance of using both taxonomic and guild approaches in order to better characterize the effects of land use change on biodiversity.

**Key words:** Cerrado, Agroecosystems expansion, Formicidae.

## Introduction

Global concern about high rates of habitat transformation has led the scientific community and political decision-makers to seek sustainable conservation strategies (DORNELAS & al. 2014, NEWBOLD & al. 2015). Tropical biodiversity, which represents two-thirds of Earth´s terrestrial biodiversity, is especially threatened (CHAZDON & al. 2009). The effects of habitat transformation are varied and include the loss or shift in diversity, and alterations to community structure and ecosystem functions and services at different spatial scales (NEWBOLD & al. 2015, SOLAR & al. 2016).

At present, many tropical regions are a mosaic of landscapes containing native vegetation fragments surrounded by different land uses, principally, for socioeconomic purposes (PERFECTO & VANDERMEER 2008, MELO & al. 2013). Land use intensification has caused local extinction of species leading to biotic homogenisations (BARLOW & al. 2007, DE LA MORA & al. 2013). Extinction of species can modify food webs and the number of biotic interactions threating the ability of ecosystems to resist to environment disturbances (MILLER & SPOOLMAN 2012). Thus, homogenous habitats produce unsustainable environments highly dependent on external inputs (PHILPOTT & ARMBRECHT 2006; PACHECO & al. 2013).

Investigations into the effects of land use change on tropical biodiversity have traditionally had a taxonomic focus (i.e. species richness and diversity) with much less attention paid to community structure (i.e. trophic structure) and functional diversity (DUYCK & al. 2011, GIBB & CUNNINGHAM 2011, HEVIA & al. 2016). Classification of assemblages into guilds (species that exploit the same type of resource in a similar way) has offered a useful tool to examine changes in community trophic structure (ROOT & al. 1967, WOODCOCK & al. 2013, KWON & al. 2014). Land use change can alter resource availability and quality, resulting in changes in food webs, including trophic position, guild distribution and species abundance and composition (GIBB & CUNNINGHAM 2011, LOURENÇO & al. 2015). Alterations in guild structure, in response to anthropogenic impacts, may negatively affect ecosystem properties and processes (BRUSSAARD 1998, DUYCK & al. 2011) through the loss of some ecological interactions (i.e. mutualism), functions (i.e. nutrient cycling) and ecosystem services (i.e. pest control).

Thus, a guild approach may reveal the structure of communities and their patterns, as well as changes in resource availability in affected habitats, and is essential for fully understanding biodiversity patterns and community ecology in both native and converted habitat (CORBELLI & al. 2015, LU & al. 2016). Many studies have shown that high-intensity

disturbance with habitat structure and resource distribution alterations may negatively affect both guild and taxonomic structure in similar way (PACHECO & al. 2009, KWON & al. 2014, CUAUTLE & al. 2016), highlighting the importance of examining the taxonomic response in parallel with the guild structure response in studies evaluating land use change impacts.

Possibly because many land use changes studies in the tropics focus on tropical forests, more attention has been paid to tree loss than afforestation (gain of trees). Consequently, we know comparably little about the effects of tree loss and afforestation on taxonomic change and guild structure across a range of vegetation types. This lack of information makes it very difficult to improve management practices for biodiversity conservation in different vegetation types (MOUILLOT & al. 2013, OVERBECK & al. 2015).

The Cerrado in Brazil varies from open grassland to closed savanna-forest formations, making it an ideal biome to explore the effect of habitat conversion as a result of either tree loss or afforestation. Furthermore, the Cerrado biome represents the second largest Brazilian biome and richest tropical savanna in the world, and is classified as a biodiversity hotspot (KLINK & MACHADO 2005). Despite this, agroecosystems have been the major human activities responsible for high rates of habitat loss and transformation in the Cerrado over the past few decades (OVERBECK & al 2007, SANO & al. 2010, GRECCHI & al. 2014).

In this study, we focus on ants because they are diverse, abundant, can be easily grouped into guilds (e.g. omnivorous, specialist predators) and respond rapidly to habitat changes (SILVESTRE & al. 2003, PACHECO & al. 2013, SOLAR & al. 2016). Ants represent a major component of tropical terrestrial biodiversity accounting for up to 80% of animal biomass (HÖLLDOBLER & WILSON 1990), and play a variety of critical functional roles such as nutrient cycling and seed dispersal (DEL TORO & al. 2012). Furthermore, ants consume a wide variety of resources including honeydew, plant exudates, animal tissue and seeds (DEL TORO & al. 2012).

Here, we evaluate the effects of land use change (afforestation and tree loss) on both the taxonomic and guild structure of ants in a Brazilian savanna (Cerrado). We examined if taxonomic and guild responses to different land use change are similar, testing if afforestation and tree loss affect species and guild frequency of occurrence distribution and composition in similar way. We expected the similar response between species and guild with greatest difference in species and guild frequency of occurrence distribution and composition at sites that had experience the greatest change in habitat structure attributes: afforested sites (by *Eucalyptus* plantation) which were previously grassland or savanna, and on deforested sites (by planted pasture) which were previously savanna-forest. We also examined how habitat

specialists and generalists (for both species and guilds) respond to land use changes, expecting that land use change would affect habitat specialists and generalists, whether species or guild, in the same way.

## Methods

### Study site

Sampling was carried out from January to March 2014 at sites around the cities of Itutinga (21°25′39.9″S 44°34′27.4″W), Itumirim (21°13′55.7″S 44°48′39.3″W) and Boa Esperança (21°04′16.8″S 45°36′36.6″W) in south Minas Gerais state, southeast Brazil. This is a transitional area between Cerrado and Atlantic Rainforest biomes (IEF 2015) characterized by a tropical climate with dry winters (April to September) and wet summers (October to March). The altitude varies between 780 to 1045 m above sea level; precipitation and mean annual temperature are 1500 mm per year and 20ºC respectively.

We conducted the field surveys on privately owned farms in three Cerrado vegetation types: C*ampo limpo*, hereafter grassland with few or no woody species; *Cerrado sensu stricto*, hereafter savanna with grasses, closed shrubs and few and scattered trees; and *Cerradão*, hereafter savanna- forest with tall trees and often complete canopy cover (SILVA & BATES 2002, OLIVEIRA-FILHO & RATTER 2002). All vegetation types are surrounded by pasture and *Eucalyptus* plantations. These land uses, grazing cattle for milk production, and wood and charcoal production respectively, are the most common pressures on the native cerrado fragments (SANO & al. 2010, IBGE 2012). Generally, these plantations are situated on private farms containing native cerrado fragments, surrounded by plantations with exotic species (e.g. the grass *Brachiaria decumbens* from Africa) in pasture and *Eucalyptus* spp. (Australian tree) in *Eucalyptus* plantation. *Eucalyptus* plantations were planted between 4 and 6 years ago and devoid of grass and understory plants, with height of 8-12 meters.

### Ant sampling

Ant assemblages were sampled within five native habitats, five pasture and five *Eucalyptus* plantation sites (apart from savanna n=3 and savannah-forest n=4, because there were no sites with appropriate size for installing the transect) within each broad vegetation type (grassland, savanna and savanna-forest). At each site, plastic pitfall traps were set along one 200 m transect with 10 sampling points, spaced 20 m apart. At each sampling point, we used one unbaited epigaeic pitfall trap (8 cm in diameter and 12 cm deep) filled with 200 ml solution of

water (99%), liquid soap (0.6 %) and salt (0.4%), and covered to protect against rain and sun. Pitfall traps were left open in the ground for 48 h.

We sorted the ant specimens to genus according to BACCARO & al. (2015), and then to morphospecies that were checked by T. S. R. Silva and G. Camacho (Laboratório de Sistemática e Biologia de Formigas, Universidade Federal do Paraná, Brazil) following DE ANDRADE & BARONI-URBANI (1999), LATTKE & al. (2007), MAYHÉ-NUNES & BRANDÃO (2002, 2005), and WILSON (2003). Voucher specimens were deposited in the Laboratório de Sistemática e Biologia de Formigas, Universidade Federal do Paraná, and in the Laboratório de Ecologia de Formigas' reference collection, Universidade Federal de Lavras.

**Definition of guild**

ROOT (1967) first proposed the term "guild" as a group of species that exploit the same class of environmental resources in a similar way. In this definition, ROOT (1967) did not consider any functional role in the guild concept; consequently showing that ecological processes are not integral to this definition. Indeed, guild classifications are based only on resource sharing that includes, for example, food, nest sites, habitat structure, roost-sites and hosts (BLONDEL 2003).

Sometimes, guilds have been wrongly considered as functional groups by presenting some guild members with similar supposed functional roles in the assemblages (BLONDEL 2003). However, in this study, we did not consider any functionally in guild classification; we just considered guild classification in relation to resource sharing. Ant species were classified into 15 guilds (Table 2.1) following the guild classification for the Neotropics using SILVESTRE & al. (2003) and BRANDÃO & al. (2009).

**Statistical analysis**

We examined if species and guild frequency of occurrence distribution respond in a similar way to land use on sites that were afforestated (*Eucalyptus* plantation) and sites that suffered woody loss (pasture). We compared the rank of species frequency of occurrence distribution and guild rank using the paired non-parametric Kolmogorov-Smirnov test. We defined taxonomic and guild frequency of occurrence as the total number of ant species and guilds occurring (min. of 1 to max. 10) per site; we had the same sampling effort per site (10 pitfall traps per site).

In order to test if there were taxonomic and guild compositional differences between native habitats and land use changes, we used permutational analysis of variance

(PERMANOVA; ANDERSON & al. 2008) for evaluating statistical significance of groups suggested by principal coordinates analysis (PCO). For both taxonomic and guild composition we used frequency of occurrence data and Bray-Curtis dissimilarity index.

To compare how species and guild habitat specialists and generalists respond to land use changes, we used CLAM approach (CHAZDON & al. 2011) using frequency of occurrence data. CLAM is a multinomial model for classifying groups into categories from two treatments (i.e. grassland and pasture or grassland and *Eucalyptus*). Such categories are: (1) native habitat associated; (2) converted habitat associated; (3) generalists; (4) too rare to classify. This analysis was conducted using the super-majority specialization threshold ($K$ = 0.667 and $p$ = 0.005). Chazdon et al. (2011) and Bicknell et al. (2014) consider CLAM analysis to be conservative and appropriate for assemblage classifications. Using the number of ant species and guilds provided by each CLAM´s category, we calculated the proportion of this number in relation to the total number of ant species and guilds sampled in order to compare between taxonomic and guild community structures; since we collected 182 ant species and 15 guilds in total (next section). All analyses were performed in R version 3.3.0 (R CORE TEAM 2016).

## Results

Across our three Cerrado vegetation types, we recorded 182 ant species (grassland: 81; savanna: 109; savanna-forest: 127) from 46 genera and seven subfamilies. Ant species were classified into 15 guilds (grassland: 13; savanna: 14; savanna-forest: 14) with omnivorous ground dominants (OGD) the richest and most frequent guild. A complete list of ant species and guilds sampled across the gradient of Cerrado tree cover is provided in Appendix 1.

### Species and guild frequency of occurrence distribution

In the grassland and savanna, contrary to our expectation, species and guilds had different responses to land use change. In both vegetation types, species frequency of occurrence distribution differed between native habitats and both *Eucalyptus* plantation and planted pasture (Tab. 2.1). The frequency of occurrence distribution for guilds did not differ between native habitats and either land use changes (*Eucalyptus* plantation and planted pasture) (Tab. 2.1). The frequency of occurrence distribution curves and the three most frequent species and guilds in grassland and savanna are presented in the Fig. 1a, b respectively.

In savanna-forest, the taxonomic and guild responses differed only in relation to conversion to *Eucalyptus* plantation. Species frequency of occurrence distribution changed in *Eucalyptus* plantation and did not change in planted pasture in relation to savanna-forest (Tab. 2.1). At the guild level, we did not detect alteration in frequency of occurrence distribution in either then *Eucalyptus* plantation or planted pasture compared with savanna-forest (Tab. 2.1). The frequency of occurrence distribution curves and the three most frequent species and guilds in savannah-forest are presented in the Fig. 2.1c.

**Tab. 2.1** Comparison of the rank of species frequency of occurrence distribution and the rank of guild frequency of occurrence distribution in response to land use change by *Eucalyptus* plantation and planted pasture in a gradient of Cerrado tree cover. Stars (*) mean significant values ($p < 0.05$).

| **Conversion type** | **Taxonomic approach** | | **Guild approach** | |
|---|---|---|---|---|
| | ***D* value** | ***p* value** | ***D* value** | ***p* value** |
| Grassland x *Eucalyptus* | 0.33 | 0.002* | 0.38 | 0.3 |
| Grassland x planted pasture | 0.34 | 0.003* | 0.23 | 0.87 |
| Savanna x *Eucalyptus* | 0.49 | 0.001* | 0.38 | 0.34 |
| Savanna x planted pasture | 0.36 | 0.001* | 0.36 | 0.33 |
| Savanna-forest x *Eucalyptus* | 0.24 | 0.001* | 0.27 | 0.61 |
| Savanna-forest x planted pasture | 0.16 | 0.06 | 0.28 | 0.62b |

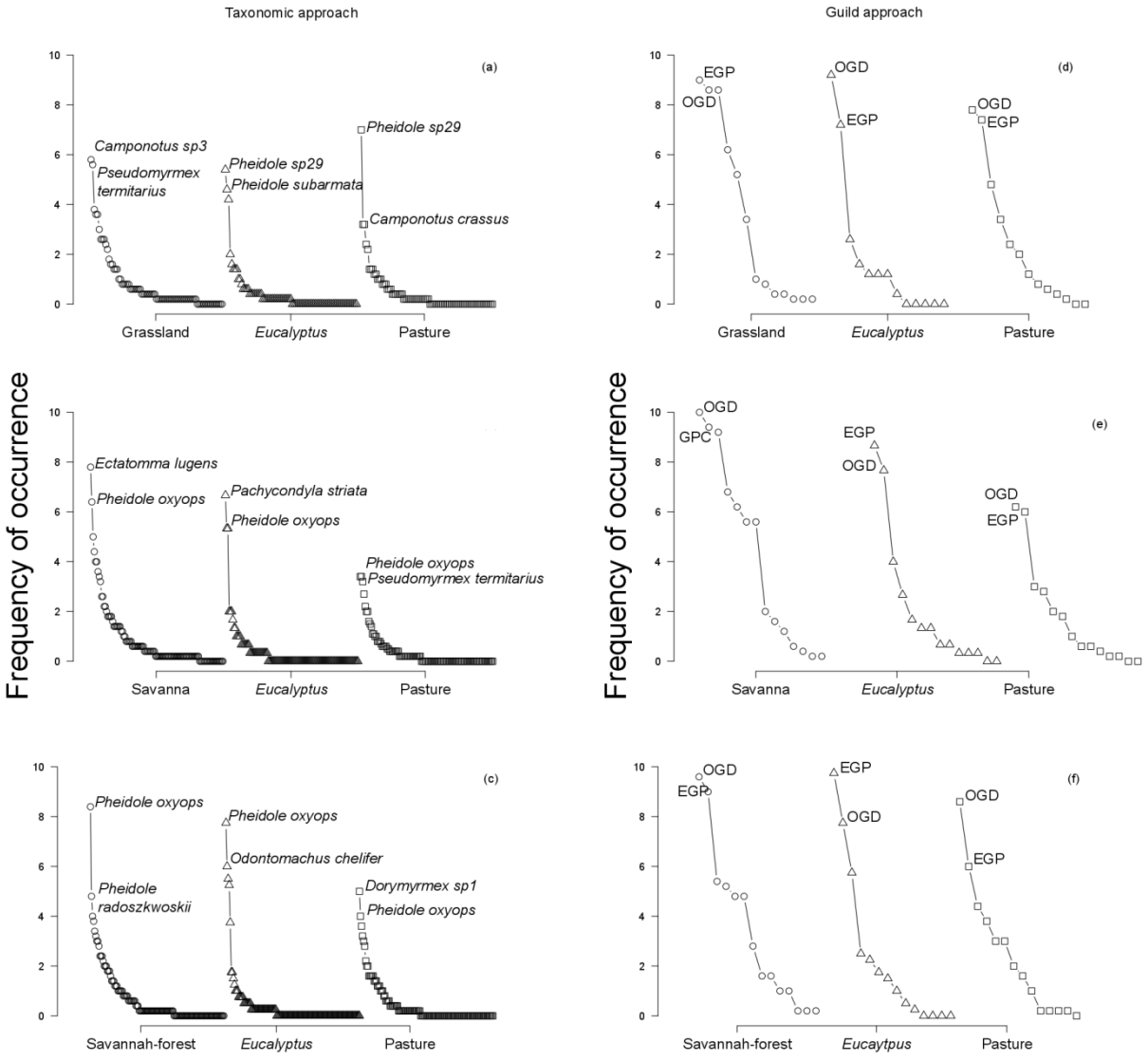


**Fig. 2.1:** Taxonomic and guild rank frequency of occurrence distribution. Results for grassland (a,d), savanna (b,e) and savanna-forest (c,f) are in first, second and third lines respectively. The three most frequent ant species and guilds are indicated in each habitat type. EGP: epigaeic generalist predator; GPC: generalist patrol camponotines; OGD: omnivorous ground dominants.

## Composition

Species and guild composition showed different responses to land use changes in grassland contrary to our hypothesis. Taxonomic composition differed between grassland and afforested areas (*Eucalyptus* plantation) and planted pasture ($F_{2,14}$ = 3.92; $p$ = 0.001) (Fig. 2.2a). By contrast, guild composition did not differ among habitats ($F_{2,14}$ = 0.83; $p$ = 0.5) (Fig. 2.2d).

In savanna and in the savannah-forest, taxonomic and guild composition had similar responses differing among all areas (savanna: taxonomic, $F_{2,12}$ = 2.63, $p$ = 0.001, Fig. 2.2b; guild: $F_{2,12}$ = 5.61, $p$ = 0.002, Fig. 2.2e) (savanna-forest: taxonomic: $F_{2,13}$ = 4.97, $p$ = 0.001, Fig. 2.2c; guild: $F_{2,13}$ = 4.70, $p$ = 0.001, Fig. 2.2f).

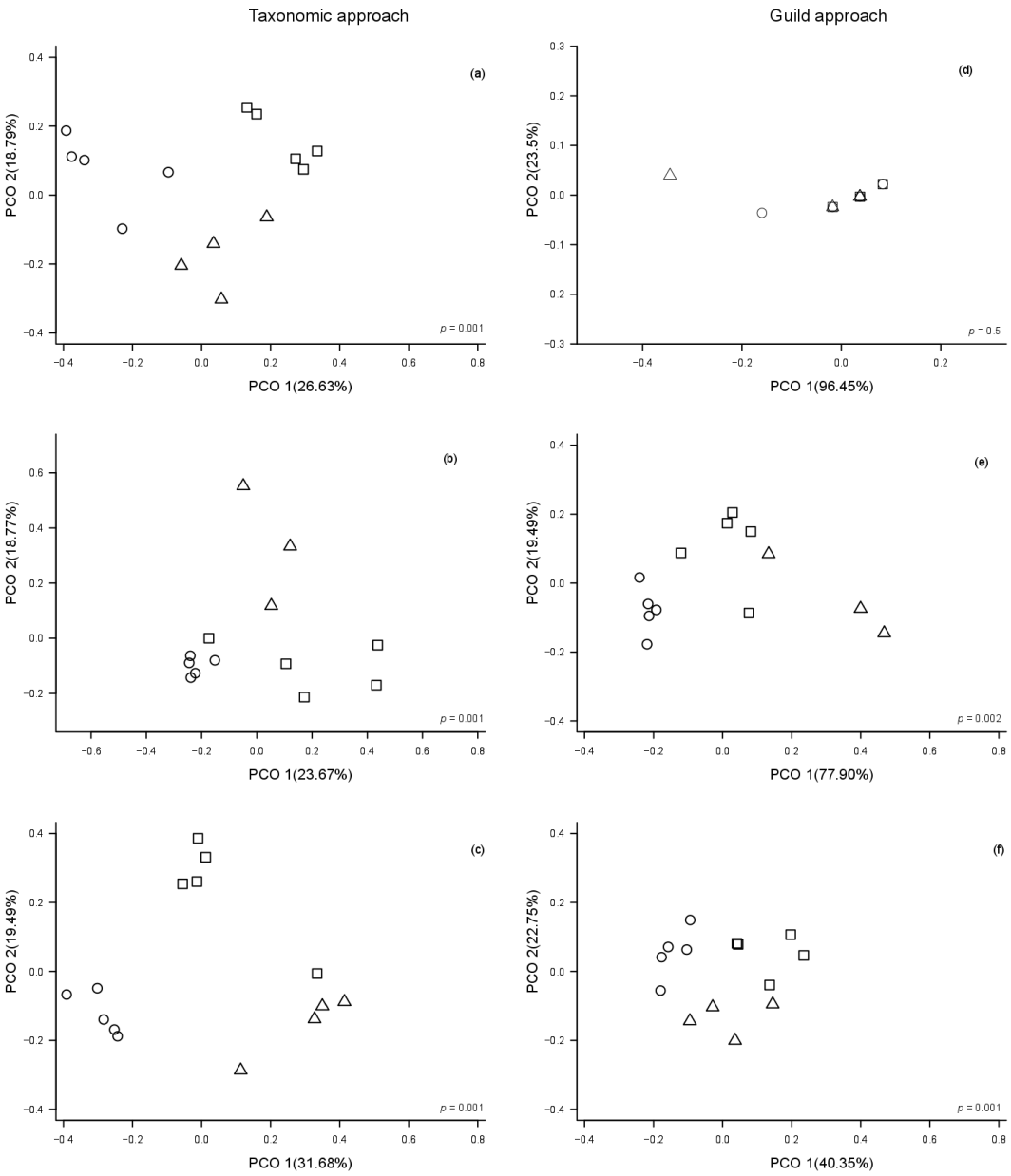


**Fig. 2.2:** Principal coordinates analysis (PCO) of taxonomic and guild composition across gradient Cerrado tree cover. Circles: native habitats; triangules: *Eucalyptus* plantation and squares: planted pasture. Grassland (a,d),savanna (b,e) and savannah-forest (c,f) are represented in first, second and third lines respectively.

**Habitat associated and generalists: species and guilds**

Species and guild habitat associated and generalists had different responses to land use change in the grassland. The comparison between grassland and *Eucalyptus* plantation had similar proportion of native habitat associated at both taxonomic and guild approaches; in the same comparison guild converted habitat associated were lost (Fig. 2.3a). Between grassland and planted pasture, we observed higher proportion of native and converted habitat associated in taxonomic than in guild approach; in the same comparison, guild native associated were lost (Fig. 2.3d). Also, there was a lower proportion of habitat generalists in taxonomic than guild approach in the conversion of grassland into both *Eucalyptus* plantation and planted pastures.

In the savanna, taxonomic and guild levels also had different responses to land use change. When comparing the savanna and both *Eucalyptus* plantation and planted pasture, we observed higher proportion of habitat associated and lower proportion of habitat generalists in taxonomic than in guild approach (Fig. 2.3b, e). We also observed a loss of guild converted habitat associated in comparison between savanna and planted pasture (Fig. 2.3e).

In savanna-forest both species and guilds produced similar response to land use change, presenting similar proportion of habitat associated at both taxonomic and guild approach when comparing savanna-forest and both *Eucalyptus* plantation (Fig. 2.3c) and planted pasture (Fig 2.3f). However, with a guild approach the habitat generalist is still higher than in taxonomic approach compared to both *Eucalyptus* plantation (Fig. 2.3c) and planted pasture (Fig. 2.3f).

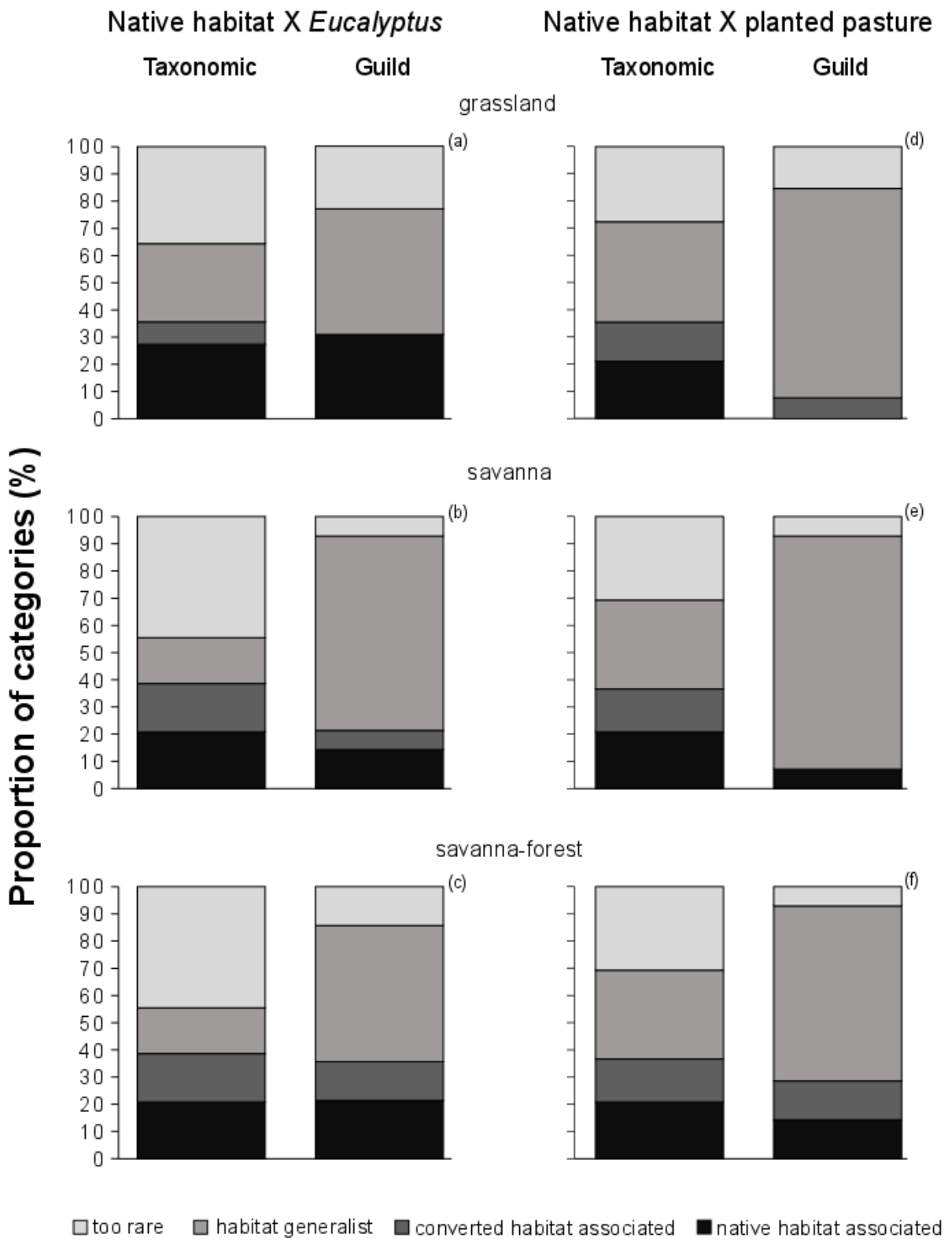


**Fig. 2.3** Comparison between taxonomic and guild approaches to land use change by *Eucalyptus* plantation and planted pasture in different Cerrado vegetation types. The X axis means the conversion type from native habitat (i.e. grassland, savanna and savanna-forest) to land use change.

## Discussion

Our study give fresh insight into the response of taxonomic and guild levels to land use change in different vegetation types; few studies have related the response of guilds of ants to land use change considering stricto sensu´s guild concept without inserting functionally in guild classification (i.e. PACHECO & al. 2009). Our findings indicate that across a gradient in tree cover (grassland to savannah-forest), ant assemblages are differently affected by land use changes both in terms of species and guilds, illustrating that studies using only one approach may provide incomplete and limited information. Such a finding is especially relevant to Cerrado that is heterogeneous in terms of vegetation type and is often lacking in studies comparing response of different biodiversity approaches to land use change. Our results demonstrate that an understanding of the impacts of land use change on assemblage structure with an unified view (i.e. exploring both taxonomic and guild approaches) is essential for predicting the effects of land use change on community dynamics.

While the impacts of land use changes at a taxonomic level have been widely documented in different tropical regions (RIZALI & al. 2012, DE LA MORA & al. 2013, DIAME & al. 2015), other components of biodiversity, such as guilds, have received less attention. Here, we demonstrate consistent alterations in taxonomic and guild community structure due to land use changes regardless of the process (afforestation or tree loss). Below, we do not discuss in detail why ant assemblage is affected by different land uses in each vegetation type (see discussion in Thesis chapter 3), rather we focus the discussion on whether the two approaches of biodiversity analysis, taxonomic and guild, respond to land use changes in a similar way.

### Species and guild frequency of occurrence distribution

We observed that land use change alters species frequency of occurrence distributions but that guild frequency of occurrence was not affected regardless of habitat transformation by either *Eucalyptus* plantation or planted pasture. In order to try explaining such finding, we briefly discuss why ant species were affected by land use change and then why guilds were not affected. The alteration in species frequency of occurrence distribution may be due to two possible explanations: (1) most species occurred less frequently compared with native habitats, or (2) dominant species occurred less frequently facilitating the presence of subordinate ones in converted habitats. Our data indicate that the conversion of native habitats in *Eucalyptus* plantation was due to the first explanation in all Cerrado vegetation types (Fig. 2.1a, b, c). In contrast, the conversion of native habitats to planted pasture in savanna and

savannah-forest likely caused dominant species to do less well (explanation 2). Changes in species frequency of occurrence with land use change (as described above) are consistent with other studies including dung beetles, ants, birds and herbaceous vegetation (ALMEIDA & al. 2011, PACHECO & al. 2013, HEVIA & al. 2016).

We found no difference in guild frequency of occurrence distribution in any Cerrado vegetation type. Guild structure appears to be more resistant and stable to conversion of native habitat into both *Eucalyptus* plantation and pasture indicating a high redundancy within guilds. Thus, the different response between ant species and guilds to land use changes indicates the higher sensitivity of species than guilds to habitat transformation during the conversion of native habitat into agroecosystems in a gradient of Cerrado tree cover.

**Composition**

We found a different response to land use change between taxonomic and guild levels in grassland, while species and guild responded in a similar way in savanna and savanna-forest. In grassland, species composition differed among different habitat types (native and converted) while guild composition was similar among different habitat types. This indicates that land use changes may lead to a loss of ant species and to a nestedness of ant species within guilds, but will not necessarily result in the loss of guilds. Such species nestedness within guilds shows that there was a systematic loss of species that make taxonomic structure more sensitive than guilds to land use change (GROC & al. 2014, DORNELAS & al. 2014, MORI & al. 2015). In grassland, similarities in guilds among different habitat types may have occurred because grassland is an original open habitat populated by species that are naturally familiarized with harsh microclimatic conditions and then presented some ant species within guilds which are less sensitive to land use changes by both *Eucalyptus* plantation and planted pasture. This result is line with PACHECO & al. (2009), BARRAGÁN & al. (2011) and PACHECO & al. (2013) that found some species from more open areas and semi-arid region are able to use converted habitats and can take advantage in getting resources available for their requirements not affecting guild or functional diversity.

In savanna and savanna forest, taxonomic and guild approaches showed similar responses with species and guild composition differing among different habitat types. Such a finding indicates that taxonomic and guild approaches are dependent each other regardless of land use change process (afforestation or tree loss). Thus, guilds will be lost when ant species are lost due to habitat transformation. The similar response of taxonomic and guild

approaches to habitat transformation is in agreement with studies using other invertebrate groups including dung beetles (CORREA & al. 2016) and spiders (CARDOSO & al. 2011). Our finding demonstrates that a combination of approaches would provide useful information to complete our understanding about anthropogenic disturbance on communities structure.

**Habitat associated and generalists: species and guilds**

In grassland and savanna, taxonomic and guild levels showed different patterns. In both vegetation types, the presence of species habitat associated in all habitat types, in both conversion to *Eucalyptus* plantation and planted pasture, shows that native and converted habitats support ant species different in their ecology; for example, species may differ in their tolerance to disturbance, and resource and microclimatic requirements (ALONSO 2010, DAHMS & al. 2010). By contrast, at a guild approach, we observed a higher proportion of habitat generalists (50-80%) than in taxonomic approach in both conversion to *Eucalyptus* plantation and planted pasture, indicating many guilds are shared among habitats. Such result indicates the shared habitat suitability preferences of some guilds, such as omnivorous ground dominants and epigaeic generalist predators guilds which can remain in stressful habitats to other ants (PACHECO & al. 2009, LU & al. 2016).

In savanna-forest, taxonomic and guild approaches revealed a similar response to land use change with similar proportion of species and guild habitat associated (native and converted habitats) in both comparisons between native habitat and both *Eucalyptus* plantation and planted pasture. This finding indicates that each habitat type harbours a particular ant assemblage and guild structure with different resource requirements. Although a guild approach showed particular guild structure, there was still a higher proportion of guild habitat generalists than species habitat associated, this finding indicates that a guild approach presents a higher number of shared, rather than specific components, among native and converted habitats and is less sensitivity at detecting land use change by both *Eucalyptus* plantation and planted pasture.

**Final considerations**

We show that land use changes have negative impacts both taxonomically and at a guild level regardless of the type of process (afforestation or tree loss). Importantly, we demonstrate that response of taxonomic diversity to land use change does not necessarily lead to the same response with guilds. We also demonstrate that taxonomic structure is more sensitive to land use change than guild structure; guild structure is more stable with higher redundancy

compared with taxonomic structure. Our study therefore suggests that taxonomic structure and guild structure are independent each other in response to land use changes.

Thus, our study offers an insight into the importance of combining two approaches to better understand the community ecology in conversion of native habitat into agroecosystems. Management and conservation of tropical savannas require a complete evaluation of response of biodiversity to land use changes impacts using different approaches. Further studies assessing land use change impacts combining multiple approaches and evaluating ecosystem functioning will be essential for the long-term management, conservation and restoration of tropical savannas.

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**Supplementary material**

**Tab. 2.S1:** List of ant species and guilds sampled in different habitat types in grassland (G), savanna (S) and savanna-forest (SF) vegetation types. E – *Eucalyptus* plantation; P – pasture; AP – arboreal predator; AS – arboreal specialist; EGP – epigaeic generalist predator; FCA – fungivorous cryptic attines; FLC – fungivorous leaf cutters; GPC – generalist patrol camponotines; GAO – ground and arboreal opportunistic; GS – ground specialist; HOS – hypogaeic omnivorous and scavengers; HS – hypogaeic specialist; HSP – hypogaeic specialist predators; L – legionary; OA – omnivorous arboreal; OGD – omnivorous ground dominants; SCP – specialist cryptic predators.

| | | Grassland | | | Savanna | | | Savanna-forest | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Ant species | Guilds | G | E | P | S | E | P | SF | E | P |
| *Ectatomma tuberculatum* | AP | | | | x | | | | | |
| *Pseudomyrmex* sp.2 | AP | | | | | | | | | x |
| *Pseudomyrmex gracilis* | AP | | | | | | | | x | |
| *Pseudomyrmex* sp.3 | AP | x | | x | x | | | | | |
| *Pseudomyrmex tenuis* | AP | | | | | x | | x | x | |
| *Nesomyrmex spininodis* | AP | | | | | | | x | | |
| *Myrmelachista* sp.1 | AS | | | | | | | x | | |
| *Myrmelachista* sp.2 | AS | x | | | | | | | | |
| *Pachycondyla harpax* | EGP | | | | | | | x | x | x |
| *Pachycondyla striata* | EGP | | x | x | x | x | x | x | x | x |
| *Neoponera verenae* | EGP | x | x | x | x | x | x | x | x | x |
| *Odontomachus chelifer* | EGP | x | x | x | | x | x | x | x | |
| *Odontomachus bauri* | EGP | x | | x | | | | x | x | x |
| *Odontomachus* sp.2 | EGP | x | | | | x | | | | x |
| *Ectatomma opaciventre* | EGP | x | x | x | x | x | | | | |
| *Ectatomma brunneum* | EGP | x | x | x | x | | x | | x | x |
| *Ectatomma lugens* | EGP | x | x | x | x | x | x | x | | x |
| *Ectatomma planidens* | EGP | x | | | x | | | | | |
| *Ectatomma edentatum* | EGP | x | x | | x | x | x | x | | |
| *Gnamptogenys sulcata* | EGP | x | | | x | | | x | x | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Gnamptogenys striatula* | EGP | x | | x | x | | x | x | | x |
| *Trachymyrmex iheringi* | FCA | | | | x | | | | | x |
| *Trachymyrmex kempfi* | FCA | | | | x | | | x | | |
| *Trachymyrmex* sp.3 | FCA | | | | x | | | | | |
| *Trachymyrmex* sp.4 | FCA | | | | x | | | | | |
| *Trachymyrmex* sp.5 | FCA | | | | x | | | | | |
| *Trachymyrmex* sp.6 | FCA | x | x | | x | | | x | | |
| *Mycocepurus* sp.1 | FCA | x | x | x | | x | | x | x | |
| *Mycocepurus* sp.2 | FCA | x | x | x | x | | | | | x |
| *Cyphomyrmex transversus* | FCA | | | | x | | x | | | |
| *Cyphomyrmex* sp.1 | FCA | | x | | x | | | | x | |
| *Cyphomyrmex* sp.2 | FCA | | | | x | | | | | |
| *Cyphomyrmex lectus* | FCA | x | | | | | x | | | |
| *Cyphomyrmex* gr *rimous* | FCA | | | | | | | | x | |
| *Cyphomyrmex minutus* | FCA | | | | x | | | | | |
| *Myrmicocrypta* cf *squamosal* | FCA | | | | | | | | | x |
| *Myrmicocrypta* sp.1 | FCA | | | | x | | | x | | |
| *Apterostigma* sp.2 | FCA | | x | | x | | x | | | |
| *Apterostigma* sp.3 | FCA | | | | | | | | | x |
| *Apterostigma* sp.4 | FCA | | | | x | | | | | |
| *Sericomyrmex* sp.1 | FCA | | | | x | | | | | |
| *Mycetarotes parallelus* | FCA | | | | | | | x | | |
| *Cyatta abscondita* | FCA | | | | x | | x | | | |
| *Trachymyrmex* sp.7 | FCA | | | | x | | | | | |
| *Apterostigma* sp.5 | FCA | | | | x | | x | | | |
| *Acromyrmex crassispinus* | FLC | | | | | | | x | x | |
| *Acromyrmex landolti* | FLC | x | x | x | x | x | x | x | x | x |
| *Atta* sp.1 | FLC | | | | | | | x | | x |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Atta* sp.2 | FLC | x | x | x | x | x | x | x | x | x |
| *Pseudomyrmex* sp.4 | GAO | | | x | | | | | | |
| *Brachymyrmex* sp.5 | GAO | x | x | x | x | | | | | x |
| *Brachymyrmex* sp.1 | GAO | x | x | x | x | x | | x | x | |
| *Brachymyrmex* sp.2 | GAO | x | x | x | | | | x | | |
| *Brachymyrmex* sp.6 | GAO | | x | x | x | | x | x | x | x |
| *Brachymyrmex* sp.1 | GAO | x | | | x | x | | | | |
| *Brachymyrmex* sp.3 | GAO | x | | | | | | | | |
| *Brachymyrmex* sp.4 | GAO | | | | | | | | | x |
| *Brachymyrmex* sp.7 | GAO | | | | x | x | | x | | |
| *Ochetomyrmex semipolitus* | GAO | x | | x | x | | x | | | |
| *Camponotus melanoticus* | GPC | x | | x | x | x | x | x | | |
| *Camponotus* sp.3 | GPC | x | x | x | x | | | | | x |
| *Camponotus blandus* | GPC | x | | | x | | x | | | x |
| *Camponotus crassus* | GPC | x | x | x | x | x | x | x | x | x |
| *Camponotus* sp.13 | GPC | | | | x | | | | x | |
| *Camponotus rufipes* | GPC | x | x | x | x | x | x | x | | |
| *Camponotus renggeri* | GPC | | | | x | | x | x | x | |
| *Camponotus leydigi* | GPC | x | x | | x | | | | | |
| *Camponotus* sp.6 | GPC | x | | | x | | | | | |
| *Camponotus* sp.5 | GPC | | | | x | | | x | | |
| *Camponotus arboreus* | GPC | | | | | | | x | | |
| *Camponotus* sp.12 | GPC | | x | | | | | | | |
| *Camponotus* sp.4 | GPC | x | | | x | | | | | |
| *Camponotus* sp.14 | GPC | | | | | | | | | x |
| *Camponotus* sp.1 | GPC | x | | | x | | | | | |
| *Camponotus* sp.8 | GPC | x | | | | | | | | |
| *Camponotus novogranadensis* | GPC | | | x | x | | | x | x | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Camponotus* sp.2 | GPC | x | | | x | | | | | x |
| *Camponotus* sp.11 | GPC | | | | | | | x | | |
| *Camponotus* sp.15 | GPC | | | | | | | | x | |
| *Camponotus* sp.7 | GPC | x | | | | | | | x | x |
| *Camponotus balzani* | GPC | | | | | | | x | | |
| *Camponotus* sp.9 | GPC | | | | | | | x | | |
| *Camponotus lespesii* | GPC | | | | x | | | x | | |
| *Pseudomyrmex termitarius* | GS | x | x | x | x | | x | x | | x |
| *Pogonomyrmex naegeli* | GS | x | x | x | x | x | x | x | x | x |
| *Oxyepoecus* gr *vezenii* | HOS | | | | | | | x | | |
| *Wasmannia auropunctata* | HOS | | | | | | | x | x | |
| *Oxyepoecus browni* | HOS | | | | | x | x | | | |
| *Oxyepoecus bruchi* | HOS | | | | | | | | x | |
| *Rogeria lirata* | HOS | | | | | | | | | x |
| *Carebara urichi* | HS | | | | | | | x | | |
| *Carebara* gr *lignata* | HS | | | | | | | | | x |
| *Gnamptogenys regularis* | HSP | | | | | x | | | x | |
| *Gnamptogenys ericae* | HSP | x | x | x | x | | x | | x | x |
| *Labidus coecus* | L | | | x | x | x | x | | | |
| *Labidus praedator* | L | | | x | | | | | | x |
| *Neivamyrmex* sp.3 | L | | | | | | x | | | |
| *Camponotus atriceps* | OA | | | | | | | x | | |
| *Cephalotes pusillus* | OA | | | | x | | | x | | |
| *Cephalotes pavonii* | OA | | | | x | | | | | |
| *Cephalotes atratus* | OA | | | | x | | | | | |
| *Wasmannia* sp.1 | OA | | | | x | | x | | | |
| *Linepithema* sp.1 | OA | | | | x | x | x | | x | x |
| *Linepithema* sp.2 | OA | x | x | x | | | x | | | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Linepithema* sp.3 | OA | x | x | x | x | | | x | | x |
| *Linepithema* sp.4 | OA | x | | x | | | | | | x |
| *Tapinoma* sp.2 | OA | x | x | x | x | | | | x | |
| *Tapinoma* sp.1 | OA | x | | x | | | | | | x |
| *Tapinoma* sp.3 | OA | x | | x | x | | | x | | |
| *Crematogaster elevans* | OA | | | | | | | x | | |
| *Crematogaster* sp.1 | OA | | | | | | | x | | |
| *Crematogaster* sp.3 | OA | | | | x | | x | | | |
| *Crematogaster* sp.4 | OA | | | | | | x | | | |
| *Crematogaster* sp.5 | OA | | | | | | | x | | |
| *Crematogaster* sp.6 | OA | x | | | | | | | | |
| *Crematogaster* sp.8 | OA | | | | | | | x | | |
| *Crematogaster* sp.11 | OA | | | | x | | | | | |
| *Crematogaster* sp.12 | OA | x | x | x | x | | x | x | | x |
| *Azteca* sp.1 | OA | | | | | | | x | | |
| *Tapinoma* sp.1 | OA | | x | | | | | | | |
| *Pheidole* sp.16 | OGD | | | | | | | x | | |
| *Pheidole* sp.27 | OGD | x | | | | | | x | x | |
| *Pheidole susannae* | OGD | | | | | | x | | x | |
| *Pheidole* sp.12 | OGD | | | | | | | | x | x |
| *Pheidole aff claviscapa* | OGD | x | x | x | | | | x | | x |
| *Pheidole triconstricta* | OGD | | | | | | x | x | | |
| *Pheidole* aff *rugiceps* | OGD | | | | | | | x | x | |
| *Pheidole transversostriata* | OGD | | | | x | x | x | x | x | x |
| *Pheidole* sp.17 | OGD | | | | | | | x | | |
| *Pheidole* sp.18 | OGD | | | | | | | | | x |
| *Pheidole* sp.23 | OGD | | | | | x | | x | | |
| *Pheidole* sp.22 | OGD | x | | | | | | | | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Pheidole cavifrons* | OGD | x | x | | | | | | | |
| *Pheidole radoszkowskii* | OGD | x | | | x | | x | x | | x |
| *Pheidole aper* | OGD | | | | x | | | x | | |
| *Pheidole* sp.19 | OGD | | | | | | | | x | |
| *Pheidole subarmata* | OGD | x | x | x | x | | x | x | x | x |
| *Pheidole* sp.30 | OGD | | | | x | | | | | |
| *Pheidole* sp.28 | OGD | | | | x | | | x | | |
| *Pheidole* sp.29 | OGD | | x | x | | x | x | | x | x |
| *Pheidole* sp.1 | OGD | | | x | | | x | | | x |
| *Pheidole scapulata* | OGD | x | x | x | x | | | | | x |
| *Pheidole oxyops* | OGD | x | x | x | x | x | x | x | x | x |
| *Pheidole vafra* | OGD | x | x | x | x | | x | x | x | x |
| *Pheidole flavens* | OGD | | | | | | | | x | |
| *Pheidole* sp.4 | OGD | | | | x | | | | | |
| *Pheidole* sp.10 | OGD | | x | x | | x | | | | |
| *Pheidole* sp.25 | OGD | | | | | | | x | | |
| *Pheidole* sp.24 | OGD | | | | | | | x | | |
| *Pheidole gertrudae* | OGD | | | | | | | | x | |
| *Pheidole* sp.13 | OGD | | | | | | | x | x | |
| *Pheidole* sp.14 | OGD | | | | | | | | | x |
| *Pheidole* sp.26 | OGD | | | | x | | x | | | |
| *Pheidole* sp.21 | OGD | | | | | | | | | x |
| *Solenopsis invicta* | OGD | x | x | x | | | | x | | |
| *Solenopsis* gr *geminate* | OGD | | | | x | | x | | | x |
| *Solenopsis* sp.9 | OGD | | | | | x | x | | | x |
| *Solenopsis* sp.3 | OGD | x | x | x | x | | | | | |
| *Solenopsis* sp.4 | OGD | x | | x | x | x | | x | x | x |
| *Solenopsis* sp.8 | OGD | | | | | x | | x | | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Solenopsis* sp.7 | OGD | | | | x | | | | | |
| *Solenopsis* sp.2 | OGD | x | | x | x | x | x | x | | |
| *Solenopsis* sp.10 | OGD | x | x | x | x | x | x | | | x |
| *Solenopsis* sp.6 | OGD | x | | | x | x | | | | |
| *Solenopsis* sp.5 | OGD | | | | | | x | | | |
| *Solenopsis* sp.11 | OGD | | | x | | | | | | |
| *Solenopsis* sp.13 | OGD | | | | | | | x | | |
| *Dorymyrmex* sp.1 | OGD | x | | x | x | | | | x | x |
| *Dorymyrmex* sp.2 | OGD | x | | | | | | | x | x |
| *Dorymyrmex* sp.3 | OGD | | | | | | | | | x |
| *Pheidole* sp.32 | OGD | | | | | | | | x | |
| *Nylanderia* sp.1 | OGD | | | | | | | x | | |
| *Nylanderia* sp.2 | OGD | | | | x | x | x | x | | |
| *Dorymyrmex* sp.5 | OGD | x | | | | | | | | |
| *Anochetus* sp.1 | SCP | | | | | | | | x | x |
| *Octostruma balzani* | SCP | | | | x | | | | | |
| *Hypoponera* sp.1 | SCP | | | | | | | x | | |
| *Hypoponera* sp.2 | SCP | | | | | | | x | | |
| *Hypoponera* sp.3 | SCP | | | x | x | | | | | |
| *Strumigenys subedentata* | SCP | | | | | x | | | | |
| *Strumigenys louisianae* | SCP | x | | | | | | x | | |
| *Strumigenys eggersi* | SCP | | | | | | | x | | |

# ARTIGO 3

## Similarity in habitat attributes reduces land use change impacts on seed removal by ants

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**Similarity in habitat attributes reduces land use change impacts on seed removal by ants**

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**ABSTRACT**

Understanding the importance of habitat similarity in reducing land use change impacts on ecosystem functioning is the key to improve management strategies of agrosystems and the effectiveness of biodiversity conservation actions. Most studies explore ecosystem functioning response to land use change are within only one vegetation type and often focus only on woody less, meaning it is not possible to draw general conclusions across other vegetation types or with other land use changes (e.g. afforestation). Across a natural gradient of tree cover we examined the consequences of natural grassland, savanna and forest conversion to planted pastures of *Brachiaria decumbens* and tree plantations of *Eucalyptus* spp. on ecosystem functioning (seed removal by ants). We expected the greatest impact on seed-removal where the converted habitat attributes differed most from the native vegetation, and that changes in seed-removal would be correlated to habitat attributes variables. We found that land use changes had impact on seed removal across the tree cover gradient. However, the degree of impact was influenced by similarity in habitat attributes and was greater where there was afforestation than woody less. Herbaceous ground cover, soil hardness and tree richness were the most important habitat attributes variables correlated with the change in seed-removal. We highlight the degree of impact of land use change on the process of seed removal varies depending on vegetation type, and it is likely correlated with the type of habitat attributes change.

**Key words:** Afforestation; woody less; agroecosystems; vegetation types; Cerrado

IN TROPICAL LANDSCAPES, THE EXTENSIVE HUMAN TRANSFORMATION OF ORIGINAL HABITAT HAS CAUSED DRAMATIC EFFECTS ON BIODIVERSITY (Barlow *et al*. 2007, Gardner *et al*. 2009). Anthropogenic land use practices, including agroecosystems, are considered the main causal factor in the loss of tropical biodiversity (Melo *et al*. 2013; Salazar *et al*. 2015) and associated ecosystem functioning (Mitchell *et al*. 2015) in recent decades. Thus, the conversion of original habitats into agrosystems may represent a threat to the ecological stability and integrity of habitats, creating unsustainable systems (Nakamura *et al*. 2003, Pacheco *et al*. 2013).

There are many effects of agroecosystems; one is altered habitat attributes (an arrangement of environmental attributes variables). Habitat attributes change can occur in savannas, depending on native vegetation type, via either the loss of trees ('woody less') or the gain of trees ('afforestation' with plantations) (Bremer & Farley 2010, Veldman *et al*. 2015a). Habitat attributes change caused by woody less results in reduced woody cover and greater homogeneity than native habitats (Tews *et al*. 2004). These habitat attributes changes result in altered microclimate conditions, productivity and resource availability (Norfolk *et al*. 2012, de la Mora *et al*. 2013).

Habitat attributes change caused by afforestation, in savannas, may increase tree cover (changing light and soil condition) excluding shade tolerant species and limiting productivity of herbaceous plants (Buscardo *et al*. 2008, Parr *et al*. 2014). In this case, the native habitats increase tree cover and reduce groundwater (Veldman *et al*. 2015b, Honda & Durigan 2016) as well as having a greater physical barrier on the ground to plant germination and animal locomotion (Buscardo *et al*. 2008). Both types of habitat alteration, woody less or afforestation, can modify the relationship between biodiversity and ecological functions by changing the composition and behavior of partner species with negative consequences for

ecosystem functioning (Philpott *et al*. 2008, Zelikova & Breed 2008, Escobar-Ramirez et al. 2012, Leal et al. 2014).

However, some agroecosystems may have a more limited-effect on biodiversity where they have a similar habitat attributes to the native habitat (Braga *et al*. 2010, Frizzo & Vasconcelos 2013, Livingston *et al*. 2013). The habitat attributes can have considerable influence in maintaining species diversity, and thus may determine the community structure in agroecosystems (Lassau & Hochuli 2004, Perfecto & Vandermeer 2008, Williams *et al*. 2012). Nevertheless, there is currently insufficient information about whether agroecosystems that have habitat attributes similar to native habitats can limit the impact on ecosystem functioning. Such information is essential to improve our capacity to determine the consequences of land use change for ecosystem functioning and better manage agroecosystems for sustainability.

In this context, to study the impact of land use by habitat attributes alteration on ecosystem functioning, we focused on the Cerrado (the Brazilian savanna) and ants. The Cerrado is a mosaic of different native vegetation types, with different habitat attributes, that has been considered one of the most threatened tropical savannas in the world (da Silva & Bates 2002, Grecchi *et al*. 2014). Over the past 50 years, around half of the Cerrado has already been replaced by different land uses, with planted pasture and agriculture (mainly soybean) covering 41.6% and 11.4% of the biome respectively (Klink & Machado 2005, Grecchi *et al*. 2014).

Ants have been used successfully as bioindicators of changing habitat conditions due to their wide geographic distribution, high diversity, functional importance and sensitivity to environmental changes (Majer *et al*. 2007, Gardner 2010, Hoffmann 2010, Philpott *et al*. 2010, Ribas *et al*. 2012). Moreover, ants have essential ecological functions such as seed dispersal, which can affect the abundance, distribution, survival and colonization of plants by

improving the chances of germination (Wang & Smith 2002, Christianini & Oliveira 2010, Gallegos *et al*. 2014). However, land use changes may reduce the quality of seed dispersal by altering ant assemblage composition with loss of larger-bodied ants (Leal *et al*. 2014), decreasing the proportion of seed-removing ant species in local community and reducing the seed removal distance; this has important consequences for the chance of seeds reaching different microhabitats (Heithaus & Humes 2003).

In this study, we examined the impacts of land use change (via woody less and afforestation) on ecosystem functioning across a tree cover gradient in the Cerrado (grassland, savanna and forest). Specifically we considered: (1) the composition of seed-removing ants; (2) the proportion of seed-removing ants in epigaeic ant assemblages; (3) the body size of seed-removing ants; and (4) the percentage of seeds removed. We predicted, first, there would be a greater difference in composition, smaller proportion of seed-removing ants, fewer large-bodied seed-removing ants and lower percentage of seed removed where the converted habitat attributes differed most from native habitat. We expected the greatest difference would be between tropical grassy habitats (grassland and savanna) and *Eucalyptus* plantation due to the afforestation process, and between forest and planted pasture due to the woody less process. Second, we predicted that changes in composition and percentage of seeds removed would be correlated with habitat attributes variables.

## METHODS

STUDY SITE AND SAMPLING DESIGN.—We conducted the fieldwork on privately owned farms in three Cerrado vegetation types during the austral summer from January to March 2014 in Itutinga (21°25′39.9″N 44°34′27.4″W), Itumirim (21°13′55.7″N 44°48′39.3″W) and Boa Esperança (21°04′16.8″N 45°36′36.6″W) in south Minas Gerais state, southeast Brazil. The region is situated in a transition area between Cerrado and Atlantic Rainforest biomes (IEF

2015), with dry winters (April to September) and wet summers (October to March). The sample sites were between 780 to 1045 m asl. Rainfall averages 1500 mm per year and the mean annual temperature is 20 ºC.

The three Cerrado vegetation types represent a gradient of tree cover: *campo limpo*, hereafter grassland, with few or no woody species; *cerrado sensu stricto*, hereafter savanna, with grasses, closed shrub and scattered shrubs; and *cerradão*, hereafter forest, with tall trees and often complete tree canopy cover (Oliveira-Filho & Ratter 2002, Silva & Bates 2002). Itutinga supports grassland vegetation type, while Itumirim and Boa Esperança have savanna and forest respectively. All Cerrado vegetation types are under planted pasture and *Eucalyptus* plantation pressures.

Almost all farms in the three vegetation types contain native vegetation fragments, containing native plant species, and traditionally some farmers use them to graze their cattle. Many areas of Cerrado have been replaced by planted pasture with exotic African grass (*Brachiaria decumbens*) to increase the carrying capacity of cattle for milk production (Cunha *et al*. 2008, Sano *et al*. 2010). *Eucalyptus* plantations have also been planted on original vegetation for wood and charcoal production, and the topsoil layer tilled to prepare the ground for planting (IBGE 2012). *Eucalyptus* trees, in grassland, were 8 meters tall and were planted 4 years before sampling; in savanna and forest – 12 meters tall and planted 6 years pre-sampling. At the time of sampling, the *Eucalyptus* plantations were devoid of grass and understory plants.

Within each vegetation type (grassland, savanna and forest), we selected five native habitat sites, five planted pasture and five *Eucalyptus* plantation (apart from savanna (n = 3) and forest (n = 4) because we could not find more sites the appropriate size for installing transects) within 5 km radius, totalling 15 sites (transects) in grassland, 13 in savanna and 14

in forest. In each site we established a 200 m transect (in a total of 42 transects) distant 50 m from native/converted habitats edge, with ten sampling points 20 m from each other.

HABITAT ATTRIBUTES SAMPLING.—To describe habitat attributes, at each sampling point we measured tree canopy cover, soil hardness, litter diversity and dry weight, percentage herbaceous ground cover, and tree or shrub (in savanna) diameter (circumference at basis height, 30 cm above ground level, > 5 cm), height, richness and density. The percentage canopy cover was calculated with digital images from fish-eye lens attached to camera positioned at height 1.5 m, and we analyzed the images using the software Gap Light Analyser 2.0 (Frazer *et al*. 1999). To measure soil hardness, we used a penetrometer launched to ground at a height 1.5 m with depth marks (cm) indicating the soil strength.

We used a 25 x 25 cm quadrat for collecting all litter and counting the number of different morphospecies of leaves, branches and sticks, and after we calculated the litter diversity using Inverse Simpson index (based on Queiroz *et al*. 2013). The litter samples were oven-dried for 96 h at 60 ºC; after which they were weighed in a precision balance to quantify litter dry weight. We estimated the percentage herbaceous ground cover using a 1 x 1 m quadrant at each sampling point. Around each sampling point, we delimited a 6 x 6 m plot and recorded the number of trees or shrub (circumference at base height > 5 cm at least), tree or shrub height, diameter and density.

ANT SAMPLING.— At each sampling point on the transect, we placed ten fresh seeds of *Croton floribundus* on the ground. *Croton floribundus* (Euphorbiaceae) is a pioneer shrub species in Cerrado biome with small elaiosome-bearing seeds (4.7 x 4.4 mm) (Paoli *et al*. 1995). The seeds were protected against vertebrate consumers by wire cage (1.5 cm mesh). Seeds were put out, on the ground, at 0800 h and continuously checked every ten minutes until 1800 h;

during this time, we hand-collected all ant species observed carrying the seeds over distances greater than 30 cm from original location. We are aware that both disperser and harvester ant species are able to remove seeds; however, our study focused on only the seed removal event without discriminating the ant species hand-collected into disperser or harvester. After 24 h from commencement of the experiment, we recorded the number of seeds that had been removed at each type of habitat.

For each seed-removing ant species from each site, we selected one specimen with medium body size among all of individuals for measuring body size; in dimorphic species we used only minors workers because we did not hand-collected the major ones. We measured head length and head width and Weber's length (distance from the anterodorsal margin of the pronotum to the posteroventral margin of the mesosoma). As all measures were highly correlated (Pearson correlation: $r = 0.95$), we used Weber's length, based on Zelikova & Breed (2008), to classifying the seed-removing ants into two size categories: large (body $\geq 2$ mm) and small (body $< 2$ mm) according to median body measures (Ness et al. 2004).

After the seed removal experiment, at the same sampling points, we installed one unbaited epigaeic pitfall trap filled with 200 ml solution of liquid soap, water and salt, which was opened for 48 h. The epigaiec pitfall traps were 8 cm in diameter and 12 cm deep, and were covered to protect against rain and sun. Ant specimens were sorted to morphospecies following Baccaro *et al*. (2015). The identification was verified by T.S.R. da Silva and G. Camacho (Laboratório de Sistemática e Biologia de Formigas, Universidade Federal do Paraná) using the keys: de Andrade & Baroni-Urbani (1999), Mayhé-Nunes & Brandão (2002, 2005), Wilson (2003), Lattke *et al*. (2007).

DATA ANALYSIS.—We performed all analyses using data from each vegetation type (grassland, savanna and forest) separately and sites (represented by the transects) in each

habitat type as unit of replication. Then, firstly, we identified which converted habitat differed most from native habitats in relation to habitat attributes in each vegetation type (i.e. *Eucalyptus* plantation and grassland and savanna; and planted pasture and forest). We performed principal component analysis (PCA) ordination with Euclidean distance resemblance matrix. This was performed considering the average value for the each habitat attributes variable in each site.

Then, we used a permutational multivariate analysis of variance (PERMANOVA, Anderson 2001) through multiple paired comparisons to test differences in species composition among habitat types (native and converted habitats). We estimated the similarity with the Jaccard index based on presence/absence of species in each habitat type. Using data on the seed-removing ant species that were hand-collected in each habitat type, we checked the presence of these same species in pitfalls from the same transect, and then, we included this data for PERMANOVA analysis in order to increase its robustness. We built a NMDS ordination to visualize the differences in composition.

To determine whether there were differences in percentage of seeds removed and proportion of seed-removing ants in epigaeic ant assemblages where the converted habitats most differed from native habitats, we carried out generalized linear models (GLM) using quasibinomial distribution. The different habitat types were our explanatory variables and percentage of seeds removed and proportion of seed-removing ants the response variables. We also used GLM to assess differences in the body size of seed-removing ants between habitats using Quasipoisson distribution. The species richness of large and small ants was used as response variable and different habitat types as explanatory variables. We performed residual analysis to check for the error distribution and adequacy of the model.

To examine if the habitat attributes variables were correlated ($r > 0.7$, based on Pacheco & Vasconcelos 2012), we used Pearson correlation coefficient in order to avoiding

collinearity. Then, we performed Hierarchical partitioning (Chevan & Sutherland 1991) to evaluate if habitat attributes variables (explanatory variables) are related to variation in percentage of seed removed (response variable). The Hierarchical partitioning is a multiple regression able to calculate the independent contribution of each explanatory variable and identify the most likely one. We also tested whether habitat attributes variables are related to changes in composition of seed-removing ants using Canonical Analysis of Principal Coordinates (CAP) (Anderson & Willis 2003). CAP is a constrained ordination that allows us to choose the distance measure (we chose Jaccard) and to use more than one explanatory variable. All the analyses were performed in R version 3.3.0 (R Development Core Team 2016).

## RESULTS

In total, we observed 38 ant species removing seeds (Table 3.S1). The ant species came from 18 genera, in which *Pheidole* was the richest genus with 14 species. We observed 16 percent of all seeds being removed (native and converted habitats) in grassland vegetation type; 21.6 percent in savanna; and 22.8 percent in forest. Pitfall trapping yielded 184 ant species belonging to 44 genera; *Pheidole* and *Camponotus* were the richest genera with 25 species each one, followed by *Solenopsis* with 13 species.

GRASSLAND.—In accordance with our expectation, the habitat attributes of the *Eucalyptus* plantation showed a greater difference to grassland than planted pasture, with herbaceous ground cover differing most (Fig. 3.1a). As predicted, there was difference in composition of seed-removing ants ($F_{2,7} = 4.1$; $p = 0.008$) between *Eucalyptus* plantation and grassland (Fig. 3.2a) correlated with differences in herbaceous ground cover and soil hardness (Table 3.1).

Similar differences between grassland and *Eucalyptus* plantation were confirmed by the other parameters. We found a lower proportion of seed-removing ants in the epigaeic assemblage ($X^2_{1,12}$ = 27.76; $p < 0.001$) (Fig. 3.3a), no larger-bodied ants and fewer smaller-bodied ants ($X^2_{1,12}$ = 24.98; $p < 0.001$) in *Eucalyptus* plantation than in the grassland. We also found a lower percentage of seeds removed in *Eucalyptus* plantation than in the grassland ($X^2_{1,12}$ = 10.39; $p = 0.001$) (Fig. 3.4a); this correlated with herbaceous ground cover (Table 3.1). We did not find differences in composition ($F_{1,8}$ = 1.7; $p = 0.08$), proportion of seed-removing ants in the epigaeic assemblage ($X^2_{1,12}$ = 0.59; $p = 0.44$), both large-bodied ants ($X^2_{1,12}$= 1.45; $p = 0.68$) ants smaller-bodied ants ($X^2_{1,12}$= 0.91; $p = 0.34$), and number of seeds removed ($X^2_{1,12}$ = 1.011; $p = 0.3$) between planted pasture and grassland.

SAVANNA.—In terms of habitat attributes, both planted pasture (contrary to our expectation) and *Eucalyptus* plantation (according to our expectation due to afforestation) differed to the savanna (Fig. 3.1b). The composition of seed-removing ants differed between *Eucalyptus* plantation and savanna ($F_{2,7}$ = 4.5; $p = 0.002$) and between planted pasture and savanna ($F_{1,9}$ = 3.9; $p = 0.001$) (Fig. 3.2b). Such differences were associated with lower shrub richness (Table 3.1).

All the others parameters also revealed differences between *Eucalyptus* plantation and savanna: proportion of seed-removing ants ($X^2_{1,10}$ = 6.13, $p = 0.01$) (Fig. 3.3b), larger (we collected zero individuals) and smaller ants ($X^2_{1,10}$ = 5.55; $p = 0.04$), and the percentage of seeds removed was higher in savanna than in *Eucalyptus* plantation ($X^2_{1,10}$ = 3.70; $p = 0.04$) (Fig. 4b) correlated with soil hardness (Table 3.1). The planted pasture also showed differences to the savanna in terms of numbers of larger ($X^2_{1,10}$ = 18.11; $p < 0.001$) and smaller ants ($X^2_{1,10}$ = 13.00; $p = 0.001$). However, planted pasture showed similarity to

savanna in relation to percentage of seeds removed ($X^2_{1,10}$ = 0.97; $p$ = 0.32) in accordance with our prediction.

FOREST.—Both *Eucalyptus* plantation (in contrast to our expectation) and planted pasture (according to our expectation due to woody less) differed in habitat attributes to the forest, but the *Eucalyptus* plantation shared similarities with the forest in terms of litter weight and tree canopy cover (Fig. 3.1c). Herbaceous ground cover was the main variable responsible for the habitat attributes difference between planted pasture and forest. However, the composition of seed-removing ants in both *Eucalyptus* plantation ($F_{2,9}$ = 4.16; $p$ = 0.004) and planted pasture ($F_{1,9}$ = 4.03; $p$ = 0.001) differed with the forest (Fig. 3.2c), associated with differences in tree diameter, herbaceous ground cover, and tree richness (Table 3.1).

In contrast, the proportion of seed removing ants was similar among converted, *Eucalyptus* plantation ($X^2_{1,11}$ = 1.59; $p$ = 0.21) and planted pasture ($X^2_{1,11}$ = 0.09; $p$ = 0.75), and forest habitats (Fig. 3.3c). Both planted pasture ($X^2_{1,11}$ = 10.55; $p$ = 0.003) and *Eucalyptus* plantation ($X^2_{1,11}$ = 12.20; $p$ = 0.001) supported fewer species of large ants than the forest; the *Eucalyptus* plantation also supported lowest number of small ant species ($X^2_{1,11}$ = 7.59; $p$ = 0.02). The percentage of seeds removed did not differ between both Eucalyptus plantation ($X^2_{1,11}$ = 0.02; $p$ = 0.96) and planted pasture ($X^2_{1,11}$ = 0.80; $p$ = 0.37) to forest (Fig. 3.4c).

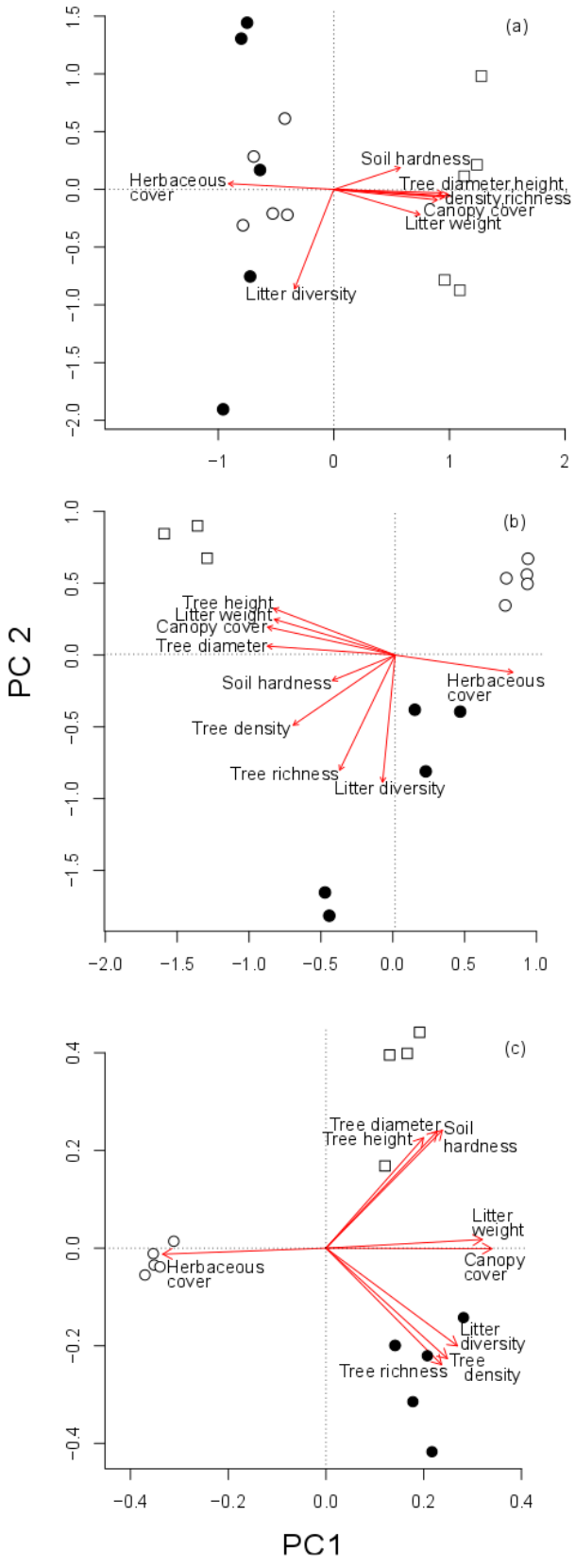


**FIGURE 3.1.** Principal Component Analysis (PCA) of habitat attributes variables of sites from native habitats (shaded dots), planted pasture (open dots) and *Eucalyptus* plantation (open squares). We used PCA to identify differences in habitat attributes among habitat types in (A) grassland, (B) savanna, and (C) forest vegetation types.

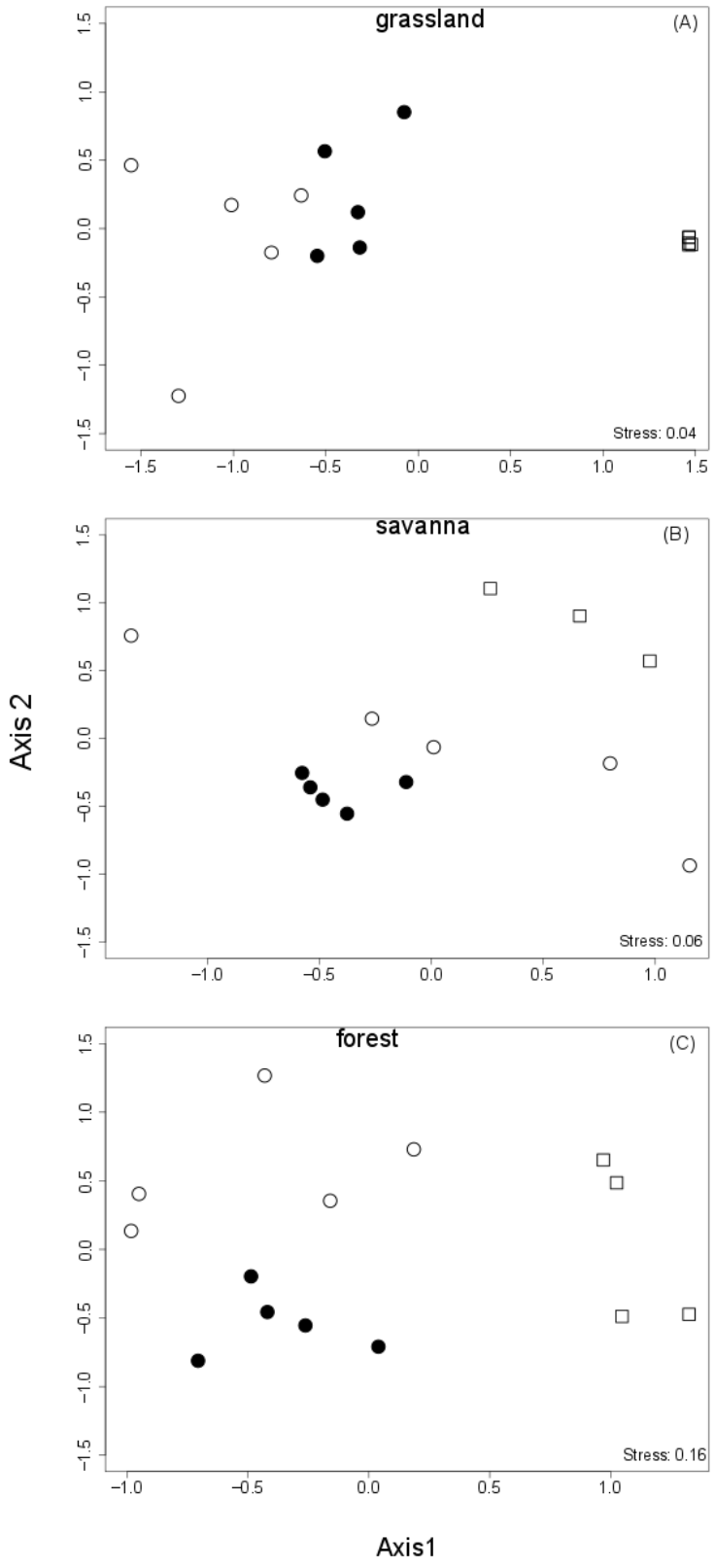


**FIGURE 3.2.** NMDS ordination showing differences in composition of seed-removing ants (Jaccard index) among native habitats (shaded dots), planted pasture (open dots) and *Eucalyptus* plantation (open squares). (A) grassland, (B) savanna, and (C) forest vegetation types.

**TABLE 3.1.** Impacts of the native habitats conversion into agroecosystems on seed-removing ant composition and seeds removed according to habitat attributes changes. In the table, only significant differences in composition and seeds removed between native and converted habitats, and significant habitat attributes influencing such differences are shown.

| Conversion type | Parameter | Habitat attributes | test value | *p*-value |
|---|---|---|---|---|
| G x E | species composition | Herbaceous ground cover | $F_{1,9} = 8.6$ | 0.001 |
| G x E | species composition | Soil hardness | $F_{1,9} = 3.4$ | 0.02 |
| G x E | seeds removed | Herbaceous ground cover | z = 1.79 | 0.04 |
| S x E | species composition | Shrub richness | $F_{1,7} = 3.6$ | 0.002 |
| S x E | seeds removed | Soil hardness | z = 1.70 | 0.04 |
| F x P and E | species composition | Herbaceous ground cover | $F_{1,9} = 2.0$ | 0.01 |
| F x P and E | species composition | Tree richness | $F_{1,9} = 3.6$ | 0.001 |
| F x P and E | species composition | Tree diameter | $F_{1,9} = 4.6$ | 0.001 |

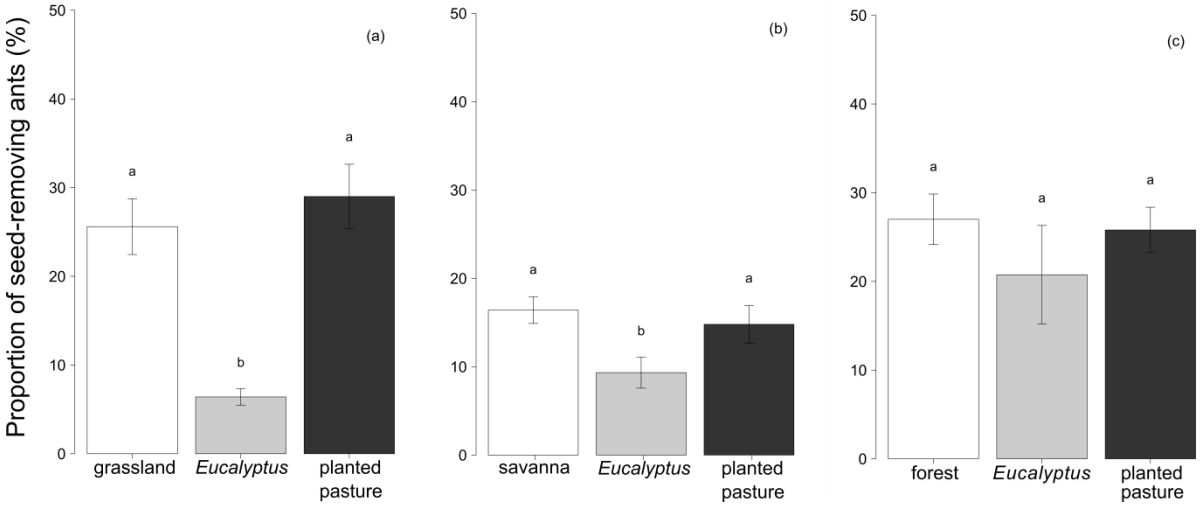


**FIGURE 3.3.** Proportion of ants that were removers in epigaeic ant assemblage among different habitats in each vegetation types: (a) grassland, (b) savanna and (c) forest. Bars indicate SEs and different letters mean significant differences ($p < 0.05$) among habitat types.

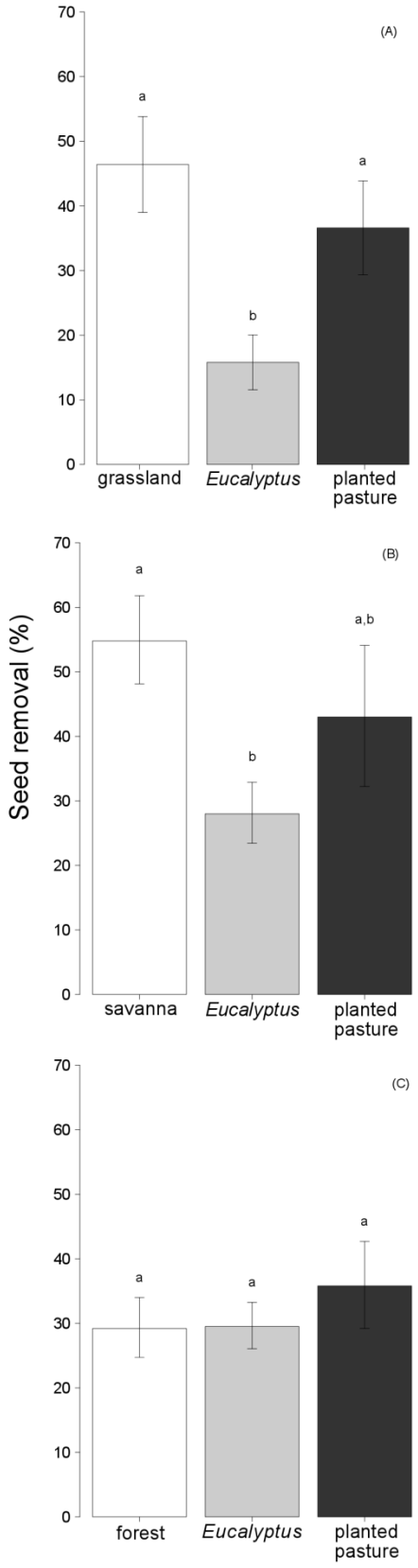


**FIGURE 3.4.** Percentage of seeds removed in different habitat types in (A) grassland, (B) savanna and (C) forest vegetation types. Bars indicate SEs and different letters indicate significant differences ($p < 0.05$) among habitat types.

# DISCUSSION

Although some studies have already reported that conversion of native habitats to agriculture and plantations leads to a decline in ecosystem functioning (e.g. Philpott & Armbrecht 2006, Zelikova & Breed 2008, Philpott *et al*. 2010, De la Mora *et al*. 2015), this study is the first, to our knowledge, to simultaneously examine the impact of land use change on ecosystem functioning by woody less and afforestation processes across a gradient of different vegetation types. Our results indicate that land use changes have impacts on seed removal by ants in all Cerrado vegetation types. However, the degree of these impacts varies depending on the land use change process (woody less or afforestation) in each vegetation type, and it is likely correlated with the type of habitat attributes change driven by particular habitat variables (e.g. herbaceous ground cover) (Fig. 3.5).

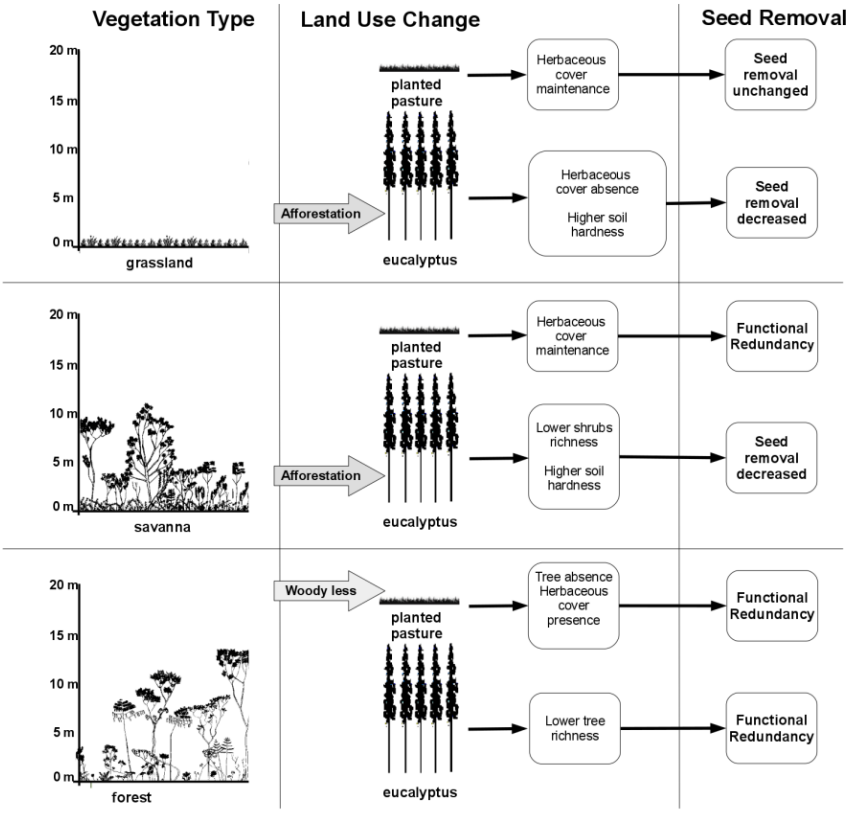


**FIGURE 3.5.** Representation of impacts of land use change by afforestation and woody less processes on seed removal across a tree cover gradient in the Cerrado. The habitat variables indicate the type of habitat attributes change influencing the land use change impacts on seed removal.

Many studies have indicated that the conversion of vegetation, that was originally open, into planted pastures, such as grasslands (e.g. *campo limpo*), has little or no impact on native fauna (e.g. Almeida *et al*. 2011, Hoffmann & James 2011, Williams *et al*. 2012) despite an obvious decline in plant species richness. Furthermore, it has been suggested that similarity in habitat attributes between original vegetation and converted land use, more broadly (i.e. not only in open habitats) could be used to evaluate the quality of converted habitat in maintaining the native fauna (Lassau & Hochuli 2004, Livingston *et al*. 2013).

In our study we show that similarity in habitat attributes between native and converted habitats helps preserve ecosystem functioning. In the grassland, there was no difference in any of the seed removal parameters between native habitat and planted pasture. In this vegetation type, a key factor underlying differences in seed removal between grassland and *Eucalyptus* plantation was the herbaceous ground cover; this was present in grassland and planted pasture, and absent in *Eucalyptus* plantation. The importance of herbaceous ground cover for seed-removing ants is in line with other reports that suggest it plays a role in determining the composition of ant assemblage by providing appropriate microclimatic conditions at ground level (Grimbacher & Hughes 2002, Williams *et al*. 2012). Furthermore, herbaceous ground cover represents important source of seeds for ants, and it can reduce the movement of seeds by rain and wind (Sprengelmeyer & Rebertus 2015), and reduces the exposure of seeds to light and desiccation.

Soil hardness differences between native sites (grassland and savanna) and *Eucalyptus* plantations were also linked to differences in ant species composition and percentage of seeds removed. Greater soil hardness in the *Eucalyptus* plantation (Fig. 2a,b) reflects the management intensity and soil compaction during the afforestation process, which seems have greater impact on ant species composition than soil hardness due to cattle trampling in pasture. It is possible that greater soil hardness reduces the availability of good nest

establishment conditions for ants, threatening the presence of both large and small seed-removing ants thereby reducing the proportion of seed-removing ants in the epigaeic assemblage, and negatively influencing the percentage of seeds removed in *Eucalyptus* plantation. Limitations in nest site availability and establishment are important factors affecting ant diversity in converted habitats (Philpott & Armbrecht 2006).

In this sense, afforestation with *Eucalyptus* plantations in both grassland and savanna vegetation types represents dramatic habitat attributes change at ground level, since we found difference in all seed removal parameters between native habitats and *Eucalyptus*. Strikingly our findings highlight the ecological importance of the herbaceous plant community in these systems; definitions of biomes and habitats must therefore move beyond simply relying on tree cover as their defining characteristic (Parr et al. 214 and Veldman et al. 2015b).

In the savanna, differences in shrub richness among savanna and converted habitats were also associated with changes on composition of seed-removing ants. Shrub richness will influence the diversity of seeds available and greater resource opportunities for seed-removing ants, thus supporting wider range of seed-removing ants. Conversely, low diversity in resources in converted habitats may have caused differences in richness of large and small seed-removing ants between native and converted habitats, consequently affecting the composition of seed-removing ants in epigaeic assemblages. The benefits of a more diverse habitat were also observed in the forest. Differences in tree richness between forest and converted habitats were also related to changes in composition reflecting in a loss of large seed-removing ants in both converted habitats. Larger seed-removing ants have been documented as more sensitive to disturbance than smaller bodied species and able to remove seeds greater distance (Leal *et al*. 2014).

In both the savanna and forest vegetation types, there was no difference in the percentage of seed removed between planted pasture and native habitats. Such a result may be

attributed to presence of herbaceous ground cover in planted pasture, which likely provides suitable microclimatic conditions (moisture and shadow at ground level) for foraging and soil-nest ants, particularly those belonging to *Pheidole* and *Solenopsis* genera. Both genera are abundant and widespread (Graham *et al*. 2004, Pacheco *et al*. 2013) and remove a large number seeds in disturbed habitats (Leal *et al*. 2014) positively contributing to seed removal. Such a finding (change in composition but no change on percentage of seeds removed) suggests a degree of functional replacement between native and planted pasture habitats in both savanna and forest vegetation types. Furthermore, *Pheidole* and *Solenopsis* (classified as small ants) are considered important genera for maintenance of soil structure in converted habitats (Sanabria *et al*. 2014), which can contribute to improve nest establishment conditions for ants.

In the forest vegetation type, tree diameter is associated with differences in composition between planted pasture and forest because it is a variable directly related to presence of trees which are lost in planted pastures due to the woody less process. The loss of woody trees alters ground layer complexity, microclimatic conditions and seed resource availability to seed-removing ants, affecting the presence of a given set of ant species (Ottonetti *et al*. 2006, Frizzo & Vasconcelos 2013), such as large-bodied seed-removing ants in this study.

Yet in the forest, a combination of similar tree-related structural variables (tree diameter, tree canopy cover and litter layer) provided greater similarity in habitat attributes between forest and *Eucalyptus* plantation than between forest and planted pasture. Such similarity decreases the negative impact of the *Eucalyptus* plantations on seed removal: although the composition of ants differed, there was no difference in the proportion of seed-removing ants (which can increase percentage of seeds removed) or in the percentage of seeds removed. Despite compositional difference, there is no change in percentage of seeds

removed between *Eucalyptus* plantation and forest indicating some degree of functional redundancy in seed-removing ants between these habitats.

It is worth highlighting that in all Cerrado vegetation types, *Eucalyptus* plantations negatively affected large seed-removing ants. Loss of large seed-removing ants may increase ant nest density, and decrease the opportunity for seeds to be transported further distance avoiding parent-offspring competition (Gómez & Espadaler 2013, Christianini & Oliveira 2010). We observed that *Eucalyptus* plantations have more twigs on the ground, deeper litter and more rocks than forests, which may decrease gap sizes and act as obstacles for larger epigaeic ants. This is one possible explanation for why we observed fewer large ants in *Eucalyptus* plantations compared to forests, however future research is necessary to confirm our hypothesis.

CONCLUSIONS AND CONSERVATION IMPLICATIONS.—We found that the impacts of habitat conversion on native fauna diversity and the ecosystem functioning in converted systems is less when there are similar habitat attributes to native habitats. We also found that afforestation (*Eucalyptus* spp. planting on native grassland and savanna) has a greater impact than woody less (planted pasture on native forests). The conversion of native forest into planted pasture leads to a small degree of functional replacement between planted pasture and forest. However, the conversion of grassland and savanna into *Eucalyptus* plantation does not follow the same pattern.

We conclude that the conversion of grassland in planted pasture has no direct negative impact on seed removal. Furthermore, in savanna given that tree richness was significantly associated with seed removal, planting scattered trees in planted pasture that was formerly savanna may be a good way of creating habitat attributes similarity to native habitats, and reduce the impact of land use change (such as already suggested by Frizzo & Vasconcelos

2013). Finally, in forest, as *Eucalyptus* plantations already share some habitat attributes variables (e.g. canopy cover and litter layer) to native habitats, reducing understory clearing (decreasing management intensity, and favoring the herbaceous ground and understory tree richness growth) could be an important action to minimize the influence on seed removal.

It is important to highlight that we are not suggesting that planted pasture is equivalent to grassland and savanna, or *Eucalyptus* plantations are equivalent to forests. Instead, we identify that these land use changes have different impacts on the maintenance and preservation of seed removal, depending on how similar habitat attributes are to the historical ecosystems. Although, most studies have given more attention to woody less process, we found greater impact of afforestation than woody less, we therefore recommend that further research and conservation attention is needed to examine the effects of grassy biomes afforestation on ecosystem functioning. Our findings contribute to understanding the impacts of land use change on native fauna, and may ultimately improve management strategies of agroecosystems in tropical landscapes, including the threatened Cerrado.

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TABLE 3.S1. Seed-removing ant species hand-collected in native and converted habitats across a tree cover gradient in the Cerrado. G – grassland; S – savanna; F – forest; E – *Eucalyptus* plantation; P – planted pasture.

| **Subfamily** | ***grassland*** | | | ***savanna*** | | | ***forest*** | | |
|---|---|---|---|---|---|---|---|---|---|
| **Ant species** | G | P | E | S | P | E | F | P | E |
| **Dolichoderinae** | | | | | | | | | |
| *Dorymyrmex* sp.1 | | | | | | X | | | |
| *Linepithema micans* Forel, 1908 | | | | | | | | X | |
| **Ectatomminae** | | | | | | | | | |
| *Ectatomma brunneum* Smith,F., 1858 | X | X | | X | X | | | | |
| *Ectatomma edentatum* Roger, 1863 | | | | X | | | X | | |
| *Ectatomma lugens* Emery, 1894 | | | | X | X | | | | |
| **Formicinae** | | | | | | | | | |
| *Camponotus rufipes* Fabricius, 1775 | X | X | | X | | | | | |
| **Myrmicinae** | | | | | | | | | |
| *Acromyrmex crassispinus* Forel, 1909 | | | | | | | | | X |
| *Atta* sp.1 | X | X | | | | | | | |
| *Atta* sp.2 | X | X | | | | | | | |
| *Atta* sp.3 | | | | | | | X | X | |
| *Atta* sp.4 | | | | | | | X | X | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| *Crematogaster obscurata* Emery, 1895 | X | X | | X | | | | | |
| *Cyphomyrmex* sp.1 | | X | | | | | | | |
| *Mycetarotes parallelus* Emery, 1906 | X | | | | | | | | |
| *Mycocepurus* sp.1 | | | | | | | X | | |
| *Pheidole oxyops* Forel, 1908 | X | X | | X | X | X | X | X | X |
| *Pheidole radoszkowskii* Mayr, 1884 | X | X | | X | | | X | X | |
| *Pheidole subarmata* Mayr, 1884 | X | X | X | X | X | | | | X |
| *Pheidole* sp.1 | | | | | | X | X | X | X |
| *Pheidole* sp.2 | X | X | | | | | | X | |
| *Pheidole* sp.3 | | | | X | | | | | |
| *Pheidole* sp.4 | | | | | | | X | | |
| *Pheidole* sp.6 | | | | | | X | X | | |
| *Pheidole* sp.7 | | | | | | | X | X | |
| *Pheidole* sp.8 | | | | | | | | X | |
| *Pheidole* sp.9 | | | | | | | | | X |
| *Pheidole* sp.10 | | | | | | | | X | |
| *Pheidole* sp.11 | | | | | | | X | | |
| *Pheidole* sp.12 | | | | | | | | X | |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| *Solenopsis* sp.1 | | | | X | | | |
| *Solenopsis* sp.2 | | | | | X | X | |
| *Trachymyrmex kempfi* Fowler, 1982 | X | | | | | | |
| *Trachymyrmex* sp.1 | | | X | | | | |
| *Wasmannia* sp.1 | | | | | X | | |
| **Paraponerinae** | | | | | | | |
| *Neoporena verenae* Forel, 1922 | X | X | | | | | X |
| *Odontomachus bauri* Emery, 1892 | | | | | X | X | |
| *Odontomachus chelifer* Latreille, 1802 | | | | | X | | |
| *Pachycondyla striata* Smith, F., 1858 | | | | | X | | |

## 4 CONCLUSÃO GERAL

Nessa tese, objetivando aumentar o conhecimento científico quanto aos impactos das mudanças no uso da terra e à conservação da biodiversidade no bioma Cerrado, eu comparei os fatores estruturando a assembleia de formigas e os impactos das mudanças no uso do solo na riqueza e composição de espécies de formigas, guildas e remoção de sementes por formigas em três fitofisionomias (campo limpo, cerrado sensu stricto e cerradão). Primeiramente eu mostro que fatores de paisagem tem muito mais influencia na assembléia de formigas em habiatts nativos que fatores locais e deveriam então ser incluídos em estratégias de conservação no Cerrado. Eu também mostro que mudanças no uso da terra afetam a assembleia de formigas indicando a importância de usar abordagens diferentes (ex. taxonômica e de guildas) para avançar no entendimento dos impactos da mudança no uso da terra. Por fim, eu destaco que o grau do impacto das mudanças no uso da terra na função ecológica de remoção de sementes varia dependento do tipo de mudança, arborização ou perda de árvores, e consequentemente da similaridade nos atributos do habitat entre agrossistemas e o habitat nativo.

Em geral, os resultados dessa tese mostram que fatores de paisagem tem importante papel na regulação da assembleia de formigas, e que essa regulação bem como o processo de remoção de sementes são negativamente afetados pelas mudanças do uso da terra nas diferentes fitofisionomias do Cerrado. Entretanto, dependendo de como a diversidade é abordada e do tipo de mudança no uso da terra (arborização ou perda de árvores) podemos obter diferentes informações sobre os impactos das mudanças no uso da terra nas comunidades biológicas. Com isso, os resultados dessa tese são valiosos para orientar avaliações de impacto de mudanças no uso da terra e estratégias de monitoramento e conservação de sistemas savânicos (ex. Cerrado), bem como, para chamar a atenção para os impactos da arborização atualmente acontecendo no Cerrado. Assim, essa tese se mostra altamente relevante para estudos futuros focados no valor de conservação e impactos das mudanças do uso da terra na biodiversidade de savanas tropicais, bem como, para guiar discussões com políticos e proprietários rurais sobre estratégias de manejo mais sustentáveis.