**Taxonomic and functional approaches reveal different responses of ant assemblages to land-use changes**

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**Abstract**

Land-use change is well documented to cause species loss. However, our understanding of the effects of land-use change on other aspects of biodiversity is still limited. We evaluated if different land-use changes (*Eucalyptus* plantation and planted pasture) affect ant species and functional groups in similar ways across three Cerrado vegetation types (grassland, savanna and savanna-forest). We found that ant species and functional responses differed with land-use change in relation to frequency of occurrence and habitat specificity and fidelity. Land-use change affected species frequency of occurrence but not functional groups, indicating that species are more sensitive than functional groups to habitat transformation. Native habitats had different indicator species and functional groups compared with converted habitats. However, we did not find functional group indicators of converted habitats in any vegetation type; indicating that there is no specificity and fidelity of functional group to converted habitats and that such an approach is less sensitive to land-use changes. In savanna and savanna-forest, species and functional groups showed the same response in relation to composition with differences between native and converted habitats. Thus, functional groups will be lost when ant species are lost. In grasslands, functional group composition was similar between native and converted habitats indicating a turnover of species within functional groups. We demonstrate that both *Eucalyptus* plantation and planted pasture affect ant species and functional groups in different ways, with negative impacts both taxonomically and less so functionally. Therefore, we show that the two aspects of biodiversity can respond independently to land-use changes and, hence, the importance of using both taxonomic and functional group approaches to evaluate the effects of land-use change on biodiversity in savanna systems.

**Keywords:** Land-use changes, Cerrado, Ants, Functional groups.

**Introduction**

Global concern about high rates of habitat transformation has led the scientific community and political decision-makers to seek sustainable conservation strategies (Dornelas et al., 2014; Newbold et al., 2015; Barlow et al., 2018). Tropical biodiversity, which represents an overwhelming majority of Earth´s biodiversity, is especially threatened by habitat transformation (Barlow et al., 2018). 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 et al., 2015; Solar et al., 2016).

 Habitat transformation has led many tropical regions to become a mosaic of landscapes containing native vegetation fragments surrounded by different land-uses, principally, for socioeconomic purposes (Melo et al., 2013). Land-use intensification has caused local extinction of species leading to biotic homogenisation (Barlow et al., 2007; De la Mora et al., 2013), the alterations of food webs and biotic interactions threatening the ability of ecosystems to resist environmental disturbances (Miller & Spoolman, 2012). Thus, degraded habitats produce unsustainable environments highly dependent on external inputs (i.e. fertilizer, pesticides) (Philpott & Armbrecht, 2006; Pacheco et al., 2013).

 Land-use change can also alter resource availability and quality, resulting in changes in food webs, functional traits and species abundance and composition (Díaz et al., 2007; Lourenço et al., 2015). Some studies have shown that both taxonomic and functional metrics can present different responses depending on intensity of disturbance and the characteristics of both native and converted habitats providing complementary insights into anthropogenic impacts (Sterk et al., 2013; Corbelli et al., 2015). Even so, investigations into the effects of land-use change on tropical biodiversity have traditionally had a taxonomic focus (i.e. species richness and diversity) with less attention paid to community structure (i.e. trophic structure) and functional diversity (Duyck et al., 2011; Hevia et al., 2016; Beiroz et al., 2018).

To better understand biodiversity patterns and consequences of land-use change, the study of functional diversity offers a useful tool to evaluate the impact of land-use change on species with particular functional traits (Robson et al., 2009; Beiroz et al., 2018). Changes to functional diversity in response to anthropogenic impacts may lead to a loss of some ecological interactions (e.g., mutualisms), functions (e.g., nutrient cycling) and ecosystem services (e.g., pest control) (Brussaard, 1998; Duyck et al., 2011; Moranz et al., 2013). Thus, a functional approach may provide better insight into the structure and ecological patterns of communities, and may be essential for fully understanding biodiversity patterns and community ecology in both native and converted habitats (Corbelli et al., 2015; Gianuca et al., 2016; Lu et al., 2016).

 Possibly because many land-use change studies in the tropics focus on tropical forests, more attention has been paid to tree loss (deforestation) than afforestation. Consequently, we know comparably little about the effects of afforestation versus tree loss on taxonomic change and functional 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 et al., 2013; Overbeck et al., 2015).

 To explore the effect of converting different native habitats as a result of either afforestation (*Eucalyptus* plantation) or tree loss (planted pastures), we focus on the Brazilian savanna (Cerrado). The Cerrado is an ideal biome because it varies from open grassland to closed savanna-forest formations. Furthermore, this 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, widespread agriculture has resulted in high rates of habitat loss and transformation in the Cerrado over the past few decades (Overbeck et al., 2007; Sano et al., 2010; Grecchi et al., 2014).

 In this study, we focus on the effects of land-use change on ants because they are diverse, abundant, can be easily grouped into functional groups and respond rapidly to habitat change (Silva & Brandão 2010; Solar et al., 2016). Ants represent a major component of tropical terrestrial biodiversity accounting for up to 80% of animal biomass (Hölldobler & Wilson 1990). They play a variety of critical functional roles such as nutrient cycling and seed dispersal, and consume a wide variety of resources including honeydew, plant exudates, animal tissue and seeds (Del Toro et al., 2012; Griffiths et al. 2018).

Here, we evaluate how land-use change (afforestation and tree loss) affects the taxonomic and functional structure of ants in the Cerrado across different native habitats. We predict that converted habitats that retain some structural similarities to natural habitat will be most resistant to change in ant communities (Filloy et al., 2010; Rabello et al. 2018; Queiroz et al., 2020). We expect a similar response between species and functional groups with declines in frequency of occurrence distribution and alteration in composition only at sites that had experienced the greatest change in habitat attributes. We consider native grassland and savanna sites that have undergone afforestation, and savanna-forest sites that have undergone tree loss to planted pasture to represent the greatest habitat change. We also have compared whether there are both species and functional groups identified as characteristics of different habitats (native or converted), in order, to better understand alterations in ant assemblages in response to land-use changes.

**Materials and methods**

**Study site**

 Sampling was carried out from January to March 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, 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 20 ºC, respectively.

We conducted the field surveys on privately owned farms in three Cerrado vegetation types: C*ampo limpo*,hereaftergrassland with few or no shrubs and no trees; *Cerrado sensu stricto*, hereafter savanna with grasses 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 planted pasture and *Eucalyptus* plantations; the most common pressures on the native Cerrado fragments (Sano et al., 2010; IBGE 2012). Generally, both planted pastures (with exotic grass *Uruclhoa decumbens* from Africa) and *Eucalyptus* plantations (with *Eucalyptus* spp.; Australian trees) are situated on private farms containing native Cerrado fragments. Pastures were planted approximately 30 years ago, and *Eucalyptus* plantations were planted between 4 and 8 years ago, are devoid of grass and understory plants, with height of 8-12 m.

**Ant sampling**

In grassland habitat, ant assemblages were sampled within five native sites, five planted pasture sites and five *Eucalyptus* plantations sites. In savanna habitat, we sampled ants within five native sites, five planted pastures sites and three *Eucalyptus* plantations sites. In savanna-forest habitat, we sampled ants in five native sites, five planted pastures sites, and four *Eucalyptus* plantations sites. Thus, overall, we sampled ant assemblages in a total of 42 sites.

At each site, 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 with a total of 420 pitfall traps: 150 in grassland, 130 in savanna and 140 in savanna-forest. Each pitfall trap (8 cm in diameter and 12 cm deep) was 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 based on the ALL protocol proposed by Agosti and Alonso (2000).

We sorted the ant specimens to genus according to Baccaro et al. (2015), and then to morphospecies that were checked by Thiago Silva and Gabriela Camacho (Laboratório de Sistemática e Biologia de Formigas, Universidade Federal do Paraná) following De Andrade and Baroni-Urbani (1999), Lattke et al. (2007), Mayhé-Nunes and 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 functional groups**

Ant species were classified into functional groups using traits (measurable features of organisms that can influence their ecology and performance; McGill et al., 2006) following a functional group classification for the Neotropics (Silvestre et al. (2003) and Silva and Brandão (2010)). We used, specifically, behavioural traits related to trophic position and foraging type. The trophic position traits used were diet (predator, fungivorous and omnivorous) and diet specificity (specialist and generalist). While, the foraging type traits used were microhabitat use (arboreal, epigaeic/ground and hypogaeic), microhabitat specificity (specialist, generalist and opportunist) and foraging behaviour (cryptic, scavengers, patrol and legionary).

Thus, ant species were classified into 15 functional groups (Appendix A) and this classification was checked by Rogerio Silva (Laboratório de Morfologia e Ecologia Funcional de Formigas, Museu Paraense Emílio Goeldi).

**Statistical analyses**

To evaluate if species and functional group frequency of occurrence distributions responded in a similar way to different land-use changes, we performed the Kolmogorov-Smirnov test for two samples. We used frequency of occurrence distributions instead of abundance values because, for ants, the raw count of the number of workers does not correspond to the natural units of diversity and may dramatically overestimate the abundance of colonies of some ant species (Gotelli et al. 2011).

Thus, this test allowed us to evaluate the species and functional group frequency of occurrence distributions between native and each converted habitat separately. We defined species and functional group frequency of occurrence as the total number of occurrence of ant species and functional groups per site (min. of 1 to max. 10); we had the same sampling effort among different sites (10 pitfall traps per site). To avoid zero-inflation, we removed the components (species or functional groups) that were absent in all habitat types. We also adjusted the *p*-values using Holm-Bonferroni methods for multiple comparisons.

To test if there were species and functional group compositional differences between native habitats and land-use changes, we used permutational analysis of variance (PERMANOVA; Anderson et al. 2008), with *adonis* function in the *vegan* package (Oksanen et al. 2019), to determine the statistical significance of groups suggested by principal coordinates analysis (PCO). For both species and functional groups composition we used frequency of occurrence data and Bray-Curtis dissimilarity.

To determine whether there were species or functional groups characteristic of particular habitats, we used an indicator value analysis (IndVal; Dufrêne & Legendre 1997). IndVal combines measurements of the degree of species specificity (uniqueness to a particular habitat) and fidelity (frequency of occurrence in a particular particular). Indicator values for each species and functional group were assigned as a result of the Monte Carlo test with 999 permutations of plots among habitat types using labdsv package (Roberts 2019).

All analyses were performed using R software version 3.6.2 (R Development Core Team 2019).

**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 functional groups(grassland: 13; savanna: 14; savanna-forest: 14) with omnivorous ground-dwellers (OGD) the richest and most frequent functional group. A complete list of ant species and functional groups sampled across the gradient of Cerrado tree cover is provided in Appendix A.

**Species and functional group frequency of occurrence distributions**

Across all vegetation types the response of ants to land-use change differed depending on whether the approach was taxonomic or functional, contrary to our expectation. In all vegetation types, species frequency of occurrence distributions, but not functional groups, differed between native habitats and converted habitats (*Eucalyptus* plantations and planted pasture) (Table 1). The frequency of occurrence distribution curves and the three most frequent species and functional groups in grassland, savanna and savanna-forest are presented in Fig. 2.

**Species and functional group composition**

Species and functional group composition showed different responses to land-use changes in grassland, contrary to our hypothesis. Species composition differed between grassland and both *Eucalyptus* plantations and planted pastures (F2,14 = 3.92; P = 0.001) (Fig. 1A). By contrast, functional group composition did not differ among these habitats (F2,14 = 0.90; P = 0.5) (Fig. 1D).

In savanna and in the savanna-forest, species and functional groups responded similarly to land use change differing in composition among all habitats types (native and converted) (savanna: species, F2,12 = 2.56, P = 0.002, Fig. 1B; functional groups: F2,12 = 7.70, P < 0.001, Fig. 1E) (savanna-forest: species: F2,13 = 4.10, P < 0.001, Fig. 1C; functional groups: F2,13 = 4.28, P < 0.001, Fig. 1F).

**Species and functional groups as indicators**

In the grassland, we detected indicator ant species only with a taxonomic approach. We found one species as an indicator of grassland and two species as indicators of *Eucalyptus* plantations (Table 2). IndVal did not select any functional group as an indicator of habitat type.

In the savanna, we found indicator species in all habitat types: savanna (eight), planted pastures (two) and *Eucalyptus* plantations (one) (Table 2). We just found one functional group (ground specialists) characteristic of a habitat; in this case, characteristic of savanna (Table 2).

In savanna-forest, we found species characteristic of the savanna-forest (six), planted pastures (one) and *Eucalyptus* plantations (three) (Table 2). We found functional groups assigned to two habitat types: savanna-forest (specialist cryptic predators, and hypogaeic omnivorous and scavengers) and *Eucalyptus* plantations (ground specialists) (Table 2).

**Discussion**

Our study gives fresh insight into the response of species and functional groups to land-use change in different vegetation types. Our findings indicate that across a gradient of tree cover (grassland to savanna-forest), ant species and functional groups are affected differently by land-use changes. Our findings are especially relevant to heterogeneous habitats, like the Cerrado, which often lack studies comparing biodiversity responses to different land-use changes. Our study demonstrates that it is important to analyse the impacts of land-use change on assemblage structure in a comprehensive way (i.e. exploring both taxonomic and functional approaches) for predicting the effects of land-use change on community dynamics. Here, we show alterations in taxonomic and functional assemblage structure due to land-use changes regardless of the type of habitat transformation (*Eucalyptus* plantations or planted pastures). Below, we do not discuss in detail why ant assemblages response varies with land-use in each vegetation type (but see Queiroz et al., 2020), rather we focus on the extent to which the two approaches of biodiversity analysis, taxonomic and functional, respond (or not) in a similar way to land-use changes.

**Species and functional group frequency of occurrence distribution**

We observed that land-use change altered the species frequency of occurrence but not functional groups, regardless of the type of habitat transformation (*Eucalyptus* plantations or planted pastures). In order to describe this finding, we briefly discuss why ant species were affected by land-use change and then why functional groups were not affected. The change in species frequency of occurrence distribution may be due to two, not mutually exclusive, situations.

The first situation is that most species occurred less frequently in converted habitats compared with native habitats; for instance, in all Cerrado vegetation types, the conversion of native habitats to *Eucalyptus* plantations led to lower frequency of occurrence for in most ant species, except for few species with high relative dominance (Fig. 2). The second situation is that all ant species occurred less frequently in converted habitats, not presenting dominance in frequency as observed in native habitats; for instance, ant responses to the drastic transformation of savanna and savanna-forest to planted pasture likely resulted in a decline in frequency of occurrence of all ant species (Fig. 2). Changes in species frequency of occurrence with land-use change (as described above) are consistent with studies on other taxa and in other regions including dung beetles, ants, birds and herbaceous vegetation (Almeida et al., 2011; Pacheco et al., 2013; Hevia et al., 2016).

We found no difference in the frequency distribution for functional groups in all Cerrado vegetation types. Even when species were irregularly distributed among different habitats, as in our case (Appendix A), the species redundancy within functional groups may decrease our capacity to detect the land-use impact in savanna systems. For instance, Omnivorous ground-dwellers (OGD) was the most numerous functional group composed by 54 ant species in this study (Appendix A). Therefore, by chance alone, there is a low probability to detect a change in the frequency of occurrence in OGD because losses of species in this functional group may be compensated by the presence of other species constituting this functional group. In a similar way, the second most abundant functional group, Epigaeic generalist predators (EGP), was composed of species that were found in all habitats and vegetation types.

The different responses to land-use changes between ant species and functional groups indicate the higher sensitivity of species than functional groups to habitat transformation during the conversion of native habitat into agroecosystems across a gradient of Cerrado tree cover. Functional structure appears to be more resistant and stable to conversion of native habitats into both *Eucalyptus* plantations and pasture indicating a high redundancy within functional groups. However, we recognise that evaluating ant assemblage in terms of frequency of occurrence using epigaeic pitfall traps can have skewed the results, mainly for functional groups, underestimating the frequency of occurrence of arboreal ants for example (Vasconcelos et al. 2014).

**Species and functional groups composition**

In the grassland, we found species and functional group composition differed with land-use change, although they responded in a similar way in savanna and savanna-forest. In the grassland, species composition differed between native and converted habitats but functional group composition was similar. We note that some species such as *Myrmelachista* sp. 2 occurred only in native habitats with few shrubs (grassland) and may have contributed to the difference in species composition between native and converted habitats (Appendix A). It is important to consider that even though *Myrmelachista* sp. 2 (arboreal ant) was collected in grassland native habitats, it is known that this species often forages on the ground and also creates satellite nests in twigs on the ground (Castro et al. 2017; Fernandes et al. 2018).

Furthermore, the difference in species composition, but not in functional groups, between native and converted habitats indicates that while land-use changes may lead to a loss of ant species and to a turnover of ant species within functional groups, it will not necessarily result in the loss of functional groups. Such species turnover within functional groups demonstrates that there was a systematic loss of species that makes taxonomic structure more sensitive to land-use change than functional structure (Dornelas et al., 2014; Groc et al., 2014; Mori et al., 2015).

In the grassland, similarities in functional groups between native and converted habitats may have occurred because grassland is an original open habitat populated by species that can cope with more extreme microclimatic conditions and may also harbour functional groups which are less sensitive to land-use change to both *Eucalyptus* plantations and planted pastures. This result is in line with Pacheco et al. (2009, 2013) and Barragán et al. (2011) who found no relationship between functional diversity and land-use change because some species from open areas and semi-arid regions were better able to inhabit converted habitats, possibly because they can take advantage of resources available for their requirements.

 However, in savanna and savanna-forest, both species and functional group composition differed between native and converted habitats. Such a finding indicates that species and functional groups change concomitantly regardless of land-use change processes probably because particular functional groups (i.e. fungivorous criptic attines in savanna and arboreal specialists in savanna-forest) may be highly sensitive to habitat transformation. Leal et al. (2012) found that some functional groups, such as cryptic species and specialist predators, tend to be most sensitive to disturbance-induced loss of key resources.

Thus, we believe that functional groups will be lost when ant species are lost due to habitat transformation. The similar response of species and functional groups to habitat transformation agrees with studies using ants (Carvalho et al. 2020), dung beetles (Beiroz et al. 2018; Carvalho et al. 2020) and spiders (Cardoso et al., 2011).

**Species and functional groups as indicators**

Across Cerrado vegetation types, the response of species and functional groups to habitat preference differed indicating the importance of understanding how the two different facets of biodiversity respond to different land-use changes. We characterized the response as different because, in all Cerrado vegetation types, we found species characteristic of native and converted habitats but the same did not happen in a functional approach.

In grassland, we did not detect functional groups characteristic of any habitat type. This result can reflect the similarity in the functional group among all habitat type showing that there is no specificity and fidelity of functional group to a specific habitat type. On the other hand, in savanna and savanna-forest, we detected species and functional groups characteristic of native habitats that seems to harbour particular ant assemblages in relation to converted habitats; for example species differ in their tolerance to disturbance, and resource and microclimatic requirements (Dahms et al., 2010). In addition, we observed that functional group indicators of native habitats (which represent a gradient of tree cover; see study sites section) differ among different vegetation types leading us to suspect that there is a level of environmental structure specialization of functional groups. By contrast, no functional group was characteristic of any converted habitat, except in the savanna-forest vegetation type where ground specialists were indicators of *Eucalyptus* plantations.

 The failure to detect functional groups characteristic of converted habitats, in all vegetation types, may be due to the functional groups approach having a broadscale predictive capacity in a land-use change scenario because species within a functional group can show a range of responses to an anthropogenic impact (Hoffmann & Andersen 2003). It is also possible that species, with similar functional traits, replace each other within a functional group resulting in a functional redundancy in response to disturbance (Gianuca et al. 2018). Both possibilities suggest that a functional group approach presents a higher number of shared, rather than specific components, between converted habitats and is less sensitive at detecting land-use change impacts by both *Eucalyptus* plantations and planted pastures.

**Conclusions**

We have demonstrated that land-use changes have negative impacts both taxonomically and less so functionally, regardless of the type of habitat transformation (*Eucalyptus* plantations or planted pastures). More importantly, we demonstrate that responses of species to land-use change do not necessarily lead to the same responses when considering functional groups for most of the parameters studied. We also demonstrate that taxonomic structure is more variable and sensitive to land-use change than functional structure; functional structure is more resistant with higher redundancy compared with taxonomic structure. Thus, our findings offer an insight into the relevance of considering a combination of approaches (both taxonomic and functional) to evaluate the conversion of native habitat into agroecosystems; however, it is essential to use species to really see how land-use change affects biodiversity.

The two approaches we used here can be applied to other savanna systems and organism groups in order to improve our understanding in how biodiversity respond to land-use change. Management and conservation of tropical savannas require a robust evaluation of responses of different facets of biodiversity to land-use changes. Therefore, 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.

**Acknowledgments**

We are grateful to Mayara Imata, Guilherme Alves, Rafaela Carvalho, Rafael Cuissi and Chaim Lasmar for fieldwork assistance, and Alexandre Casadei, Gabriela Camacho and Thiago Silva for taxonomic support. We thank the farmers for access to their lands, and Renata Pacheco, Paulo Pompeu, Inara Leal, Luiz Magnago, Wallace Beiroz and Livia Prado for their feedback on the manuscript previous draft. AMR and ACMQ thank Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) for the PhD scholarship (PDSE processes 4934/14-08 and 8794/14-06 respectively). CRR thanks Fundação de Amparo à Pesquisa do Estado de Minas Gerais (FAPEMIG-CRA-PPM-000243/14) for the grant.

“Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at XXXXX.”

**References**

Agosti, D. & Alonso, L.E. (2000). The ALL protocol: a standard protocol for the collection of ground-dwelling ants. In: Agosti, D.; Majer, J.D.; Tennant, A. & Schultz, T. (eds.), Standard methods for measuring and monitoring biodiversity. Smithsonian Institution Press, Washington and London, pp. 204-206.

Almeida, S., Louzada, J., Sperber, C. & Barlow, J. (2011). Subtle land-use change and tropical biodiversity: dung beetle communities in Cerrado grasslands and exotic pastures. *Biotropica*, 43, 704-710.

Anderson, M., Gorley, R. & Clarke, K.P. (2008). For PRIMER: guide to software and statistical methods. PRIMER-E, Plymouth, UK.

Baccaro, F.B., Feitosa, R.M., Fernandez, F., Fernandes, I.O., Izzo, T.J., Souza, J.L.P. & Solar, R. (2015). *Guia para gêneros de formigas do Brasil*. Manaus: INPA.

Barragán, F.; Moreno, C.E., Escobar, F., Halffter, G. & Navarrete, D. (2011). Negative impacts of human land use on dung beetle functional diversity. *Plos One*, 6, e17976.

Barlow, J., Gardner, T.A., Araujo, I.S., Avila-Pires, T.C., Bonaldo, A.B., Costa, J.E., et al. (2007). Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences*, 104, 18555-18560.

Barlow, J., França, F., Gardner, T.A., Hicks, C.C., Lennox, G.D., Berenguer, E., et al. (2018). The future of hyperdiverse tropical ecosystems. *Nature*, 559, 517-526.

Beiroz, W., Sayer, E., Slade, E., Audino, L., Braga, R.F., Louzada, J. & Barlow, J. (2018). Spatial and temporal shifts in functional and taxonomic diversity of dung beetles in a human-modified tropical forest landscape. *Ecological Indicators*, 95, 518-526.

Brussaard, L. (1998). Soil fauna, guilds, functional groups and ecosystem processes. *Applied Soil Ecology*, 9, 123-135.

Cardoso, P., Pekár, S., Jocqué, R. & Coddington, J.A. (2011). Global patterns of guild composition and functional diversity of spiders. *Plos one*, 6, e21710.

Carvalho, R.L., Andersen, A.N., Anjos, D.V., Pacheco, R., Chagas, L. & Vasconcelos, H.L. (2020). Understanding what bioindicators are actually indicating: Linking disturbance responses to ecological traits of dung beetles and ants. *Ecological Indicators*, 108, 105764.

Castro, G.H.P.; Kayano, D.Y.; Souza, R.F.; Hilsdorf, A.W.S.; Feitosa, R.M. & Morini, M.S.C. (2017). Seasonal patterns of the foraging ecology of *Myrmelachista arthuri* Forel, 1903 (Formicidae: Formicinae). Sociobiology, 64, 237-243.

Corbelli, J.M., Zurita, G.A., Filloy, J., Galvis, J.P., Vespa, N.I. & Bellocq, I. (2015). Integrating taxonomic, functional and phylogenetic beta diversities: Interactive effects with the biome and land use across taxa. *Plos one*, 15, e0126854.

Dahms, H., Mayr, S., Birkhofer, K., Chauvat, M., Melnichnova, E., Wolters, V. & Dauber, J. (2010). Contrasting diversity patterns of epigeic arthropods between grasslands of high and low agronomic potential. *Basic and Applied Ecology*, 11, 6-14.

De Andrade, M.L. & Baroni-Urbani, C. (1999). Diversity and adaptation in the ant genus Cephalotes, past and present. *Staatliches Museum für Naturkunde*, 271, 1-889.

De la Mora, A., Murnen, C.J. & Philpott, S.M. (2013). Local and landscape drivers of biodiversity of four groups of ants in coffee landscapes. *Biodiversity and Conservation*, 22, 871-888.

Del Toro, I., Ribbons, R.R. & Pelini, S.L. (2012). The little things that run the world revisited: a review of ant-mediated ecosystem services and disservices (Hymenoptera: Formicidae). *Myrmecological News*, 17, 133-146.

Díaz, S., Lavorel, S., De Bello, F., Qué, F., Grigulis, K. & Robson, T.M. (2007). Incorporating plant functional diversity effects in ecosystem service assessments. *Proceedings of the National Academy of Sciences of the United States of America*, 104, 20684-20689.

Dornelas, M., Gotelli, N.J., McGill, B., Shimadzu, H., Moyes, F., Sievers, C. & Magurran, A.E. (2014). Assemblage time series reveal biodiversity change but not systematic loss. *Scienc*e, 344, 296-299.

Dufrêne, M. & Legendre, P. (1997). Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological Monographs*, 67, 345-366.

Duyck, P.F., Lavigne, A., Vinatier, F., Achard, R., Okolle, J.N. & Tixier, P. (2011). Addition of a new resource in agroecosystems: Do cover crops alter the trophic positions of generalist predators? *Basic and Applied Ecology*, 12, 47-55.

Fernandes, T.T.; Souza-Campana, D.R.; Silva, R.R.; Morini, M.S.C. (2018). Ants that frequently colonize twigs in the leaf litter of different vegetation habitats. *Sociobiology*, 65, 340-344.

Filloy, J., Zurita, G.A., Corbelli, J.M. & Bellocq, M.I. (2010). On the similarity among birds communities: Testing the influence of distance and land use. *Acta Oecologia*, 36, 333-338.

Gianuca, A., Pantel, J.H. & de Meester, L. (2016). Disentangling the effect of body size and phylogenetic distances on zooplankton top-down control of algae. *Proceedings Royal Society B*, 283, 20160487.

Gianuca, A., Engelen, J., Brans, K.I., Hanashiro, F.T.T., Vanhamel, M., van der Berg, E.M., et al. (2018). Taxonomic, functional and phylogenetic metacommunity ecology of cladoceran zooplankton along urbanization gradients. *Ecography*, 41, 183-194.

Gotelli, N., J., Ellison, A.M., Dunn, R.R. & Sanders, N.J. (2011). Counting ants (Hymenoptera: Formicidae): biodiversity sampling and statistical analysis for myrmecologists. *Myrmecological News*, 15, 13-19.

Grecchi, R.C., Gwyn, Q.H.J., Benie, G.B., Formaggio, A.R. & Fahl, F.C. (2014). Land use and land cover changes in the Brazilian Cerrado: A multidisciplinary approach to assess the impacts of agricultural expansion. *Applied Geography*, 55, 300-312.

Griffiths, H.M., Ashton, L.A., Walker, A.E., Hasan, F., Evans, T.A., Eggleton, P. & Parr, C.L. (2018). Ants are the major agents of resource removal from tropical rainforests. *Journal of Animal Ecology*, 87, 293-300.

Groc, S., Delabie, J.H.C., Fernandez, F., Leponce, M., Orivel, J., Silvestre, R., et al. (2014). Leaf-litter ant communities (Hymenoptera: Formicidae) in a pristine Guianese rainforest: stable functional structure versus high species turnover. *Myrmecological news*, 19, 43-51.

Hevia, V., Carmona, C.P., Azcárate, F.M., Torralba, M., Alcorlo, P., Ariño, R., et al. (2016). Effects of land use on taxonomic and functional diversity: a cross-taxon analysis in a Mediterranean landscape. *Oecologia*, 181, 959-970.

Höldobler, B. & Wilson, E.O. (1990). *The ants*. Berlin: Springer.

Hoffmann B.D. & Andersen, A.N. (2003). Response of ants to disturbance in Australia, with particular reference to functional groups. *Austral Ecology*, 28, 444-464.

IBGE. Instituto Brasileiro de Geografia e Estatística. (2012). <http://www.cidades.ibge.gov.br>/ Accessed 10 April 2015.

IEF. Instituto Estadual de Florestas. (2015). http://www.ief.mg.gov.br/florestas/ Accessed 15 July 2015.

Klink, C.A. & Machado, R. B. (2005). Conservation of the Brazilian Cerrado. *Conservation Biology*, 19, 707-713.

Lattke, J.E., Fernández, F. & Palacio, E.E. (2007). Identification of the species of Gnamptogenys Roger in the Americas*. Memoirs of the American Entomological Institute*, 80, 254-270.

Leal, I.R., Filgueiras, B.K.C., Gomes, J.P., Iannuzzi, L. & Andersen, A. (2012). Effects of habitat fragmentation on ant richness and functional composition in Brazilian Atlantic forest. *Biodiversity and Conservation*, 21, 1687-1701.

Lourenço, G.M., Campos, B.F. & Ribeiro, S.P. (2015). Spatial distribution of insect guilds in a tropical montane rainforest: effects of canopy structure and numerically dominant ants. *Arthropod-Plant Interactions*, 9, 163-174.

Lu, Z., Hoffman, B.J. & Chen Y. (2016). Can reforested and plantation habitats effectively conserve SW China´s ant biodiversity? *Biodiversity and Conservation*, 25, 753-770.

McGill, B.J., Enquist, B.J., Weiher, E. & Westoby, M. (2006). Rebuilding community ecology from functional traits. *Trends in Ecology and Evolution*, 21, 178-185.

Mayhé-Nunes, A.J. & Brandão, C.R.F. (2002). Revisionary studies on the attine ant genus Trachymyrmex Forel. Part 1: definition of the genus and the opulentus group (Hymenoptera: Formicidae). *Sociobiology*, 40, 667-698.

Mayhé-Nunes, A.J., & Brandão, C.R.F. (2005). Revisionary studies on the attine ant genus *Trachymyrmex* Forel. Part 2: The Iheringi group (Hymenoptera: Formicidae). *Sociobiology*, 45, 271-305.

Melo, F.P.L., Arroyo-Rodríguez, V., Fahrig, L., Martínez-Ramos, M. & Tabarelli, M. (2013). On the hope for biodiversity-friendly tropical landscapes. *Trends in Ecology & Evolution*, 28, 462-468.

Miller, G.C. & Spoolman, S.E. (2012). *Essentials of Ecology*. Cengage Learning.

Moranz, R.A., Debinski D.M., Winkler, L., Trager, J., McGranahan, D.A., Engle, D.M. & Miller, J.R. (2013). Effects of grassland management practices on ant functional groups in central North America. *Journal of Insect Conservation*, 17, 699-713.

Mori, A.S., Shiono, T., Haraguchi, T.F., Ota, A.T., Koide, D., Ohgue, T., et al. (2015). Functional redundancy of multiple forest taxa along an elevational gradient: predicting the consequences of non-random species loss. *Journal of Biogeography*, 42, 1383-1396.

Mouillot, D., Graham, N.A.J., Villéger, S., Mason, N.W.H. & Bellwood, D.R. (2013). A functional approach reveals community responses to disturbances. *Trends in Ecology & Evolution*, 28, 167-177.

Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., et al. (2015). Global effects of land use on local terrestrial biodiversity. *Nature*, 520, 45-50.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., et al. (2019). Vegan: Community Ecology Package. R package version 2.5-6. http://cran.r-project.org/package=vegan.

Oliveira-Filho, A.T. & Ratter, J. (2002). Vegetation physiognomies and woody flora of the Cerrado biome. In Oliveira, P.S., Marquis, T.J. (Eds), *The Cerrados of Brazil: Ecology and natural history of a neotropical* savannah (pp. 91-120). New York: Columbia University Press.

Overbeck, G.E., Müller, S.C., Fidelis, A., Pfadenhauer, J., Pillar, V.D., Blanco, C.C., et al. (2007). Brazil´s neglected biome: The South Brazilian *Campos*. *Perspectives in Plant Ecology, Evolution and Systematics*, 9, 101-116.

Overbeck, G.E., Vélez-Martin, E., Scarano, F.R., Lewinsohn, T.M., Fonseca, C.R., Meyer, S.T., et al. (2015). Conservation in Brazil needs to include non-forest ecosystems. *Diversity and Distributions*, 21, 1455-1460.

Pacheco, R., Silva, R.S., Morini, M.S. & Brandão, C.R.F. (2009). A comparison of the leaf-litter ant fauna in a secondary Atlantic Forest with an adjacent pine plantation in Southeastern Brazil. *Neotropical Entomology*, 38, 55-65.

Pacheco, R., Vasconcelos, H.L., Groc, S., Camacho, G.P. & Frizzo, T.L.M. (2013). The importance of remnants of natural vegetation for maintaining ant diversity in Brazilian agricultural landscapes. *Biodiversity and Conservation*, 22, 983-997.

Philpott, S.M. & Armbrecht, I. (2006). Biodiversity in tropical agroforests and the ecological role of ants and ant diversity in predatory function. *Ecological Entomology*, 31, 369–377.

Queiroz, A.C.M., Rabello, A.M., Braga, D.L., Santiago, G.S., Zurlo, L.F., Philpott, S.M. & Ribas, C.R. (2020). Cerrado vegetation types determine how land use impacts ant biodiversity. *Biodiversity and Conservation*, 29, 2017-2034.

R Development Core Team (2019). R: A language and environment for statistical computing. *R foundation for statistical computing*, Vienna, Austria (URL http://www.R-project.org).

Rabello, A.M., Parr, C.L., Queiroz, A.C.M., Braga, D.L., Santiago, G.S. & Ribas, C.R. (2018). Similarity in habitat attributes between native and agroecosystems reduces land use change impacts on seed removal by ants. *Biotropica*, 50, 39-49.

Roberts, D.W. (2019). Labdsv: ordination and multivariate analysis for Ecology. R. package version 2.0-1. http://cran.r-project.org/package=labdsv.

Robson, T.C., Baker, A.C. & Murray, B.R. (2009). Differences in leaf-litter invertebrate assemblages between radiata pine plantations and neighbouring native eucalypt woodland. *Austral Ecology*, 34, 368-376.

Sano, E.E., Rosa, R., Brito, J.L.S. & Ferreira, L.G. (2010). Land cover mapping of the tropical savanna region in Brazil. *Environmental Monitoring and Assessment*, 166,113-124.

Silva, J.M.C. & Bates, J.M. (2002). Biogeographic patterns and conservation in the South American Cerrado: A tropical savanna hotspots. *Bioscience*, 52, 225-233.

Silva, R.R. & Brandão, C.R.F. (2010). Morphological patterns and community organization in leaf-litter ant assemblages. *Ecological Monographs*, 80, 107-124.

Silvestre, R., Brandão, C.R.F. & Silva, R.R. (2003). Grupos funcionales de hormigas: el caso de los gremios del Cerrado, Brasil. In Fernandéz, F. (Ed.), *Introducción a las hormigas de la región Neotropical* (pp. 113-148). Bogotá: Instituto de investigación de recursos biológicos Alexander von Humboldt.

Solar, R.R.C., Barlow, J., Andersen, A.N., Schoereder, J.H., Berenguer, E., Ferreira, J.N. & Gardner, T.A. (2016). Biodiversity consequences of land-use change and forest disturbance in the Amazon: A multi-scale assessment using ant communities. *Biological Conservation*, 197, 98-107.

Sterk, M., Gort, G., Klimkowska, A., van Ruijven, J., van Teeffelen, A.J.A. & Wamelink, G.W.W. (2013). Assess ecosystem resilience: linking response and effect traits to environmental variability. *Ecological Indicators*, 3**0**, 21-27.

Vasconcelos, H.L.; Frizzo, T.L.M.; Pacheco, R.; Maravalhas, J.B.; Camacho, G.P.; Carvalho, K.S.; Koch, E.B.A.; Pujol-Luz, J.R. (2014). Evaluating sampling sufficiency and the use of surrogates for assessing ant diversity in a Neotropical biodiversity hotspot. *Ecological Indicators*, 46, 286-292.

Wilson, E.O. (2003). *Pheidole in the New World: A dominant, hyperdiverse ant genus*. Cambridge: Harvard University Press.

**Legends**

**Table 1.** Comparison of the species and functional group frequency of occurrence distributions in response to land-use change by *Eucalyptus* plantations and planted pastures in a gradient of Cerrado tree cover. The D-values mean results of Kolmogorov-Smirnov statistic test for two samples. The p-values are adjusted by the Holm-Bonferroni method.

|  |  |  |
| --- | --- | --- |
| **Conversion type** | **Taxonomic approach** | **Functional approach** |
|  | ***D-*value** | ***p-*value** | ***D-*value** | ***p-*value** |
| Grassland x *Eucalyptus*  | 0.32 | **0.002** | 0.38 | 0.58 |
| Grassland x planted pastures | 0.32 | **0.043** | 0.23 | ~1.00 |
| Savanna x *Eucalyptus*  | 0.53 | **< 0.001** | 0.36 | ~1.00 |
| Savanna x planted pastures | 0.39 | **< 0.001** | 0.36 | 0.66 |
| Savanna-forest x *Eucalyptus*  | 0.31 | **0.002** | 0.29 | 0.87 |
| Savanna-forest x planted pastures | 0.18 | **< 0.001** | 0.29 | 0.66 |

**Table 2.** Ant species and functional groups as indicators of different habitat types (native and converted habitats) in the three Cerrado vegetation types. IndVal: indicator value. *p* means probability as a result of permutation test. The table only presents the significant results of IndVal analysis (*p* < 0.05).

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Approach** |  | **IndVal** | ***p***  | **Habitat type** |
| Grassland: |  |
| **Taxonomic** | *Brachymyrmex* sp. 2 | 1 | 0.002 | Grassland |
| *Dorymyrmex* sp. 1 | 0.71 | 0.021 | *Eucalyptus* |
| *Pachycondyla striata* | 0.64 | 0.046 | *Eucalyptus* |
| Savanna: |  |  |  |  |
| **Taxonomic** | *Brachymyrmex* sp. 5 | 1 | 0.003 | Savanna |
| *Ochetomyrmex semipolitus* | 1 | 0.001 | Savanna |
| *Camponotus blandus* | 0.83 | 0.013 | Savanna |
| *Camponotus* sp. 2 | 0.80 | 0.035 | Savanna |
| *Camponotus* sp. 4 | 0.80 | 0.031 | Savanna |
| *Pheidole radoszkowskii* | 0.62 | 0.018 | Savanna |
| *Pheidole vafra* | 0.62 | 0.012 | Savanna |
| *Pseudomyrmex termitarius* | 0.50 | 0.004 | Savanna |
| *Pheidole* sp. 29 | 0.83 | 0.012 | Planted pastures |
| *Solenopsis* sp. 8 | 0.67 | 0.037 | Planted pastures |
| *Pheidole triconstricta* | 0.80 | 0.031 | *Eucalyptus* |
| **Functional group** | Ground specialists | 0.50 | 0.004 | Savanna |
| Savanna-forest: |  |  |  |  |
| **Taxonomic** | *Pheidole* sp. 17 | 1 | 0.001 | Savanna-forest |
| *Camponotus lespesii* | 1 | 0.002 | Savanna-forest |
| *Ectatomma edentatum* | 0.80 | 0.015 | Savanna-forest |
| *Nylanderia* sp. 1 | 0.80 | 0.014 | Savanna-forest |
| *Wasmannia auropunctata* | 0.80 | 0.014 | Savanna-forest |
| *Pheidole radoszkowskii* | 0.64 | 0.033 | Savanna-forest |
| *Camponotus renggeri* | 0.59 | 0.042 | Planted pastures |
| *Camponotus* sp. 3 | 0.80 | 0.012 | *Eucalyptus* |
| *Dorymyrmex* sp. 3 | 0.80 | 0.013 | *Eucalyptus* |
| *Pseudomyrmex termitarius* | 0.65 | 0.004 | *Eucalyptus* |
| **Functional group** | Specialist cryptic predators | 0.69 | 0.021 | Savanna-forest |
| Hypogaeic omnivorous and scavengers | 0.64 | 0.024 | Savanna-forest |
| Ground specialists | 0.62 | 0.005 | *Eucalyptus* |

**Fig. 1.** Principal coordinates analysis (PCO) of species and functional group composition across a gradient of Cerrado tree cover. Circles: native habitats; triangles: *Eucalyptus* plantations; squares: planted pastures. Grassland (A,D), savanna (B,E) and savanna-forest (C,F) in first, second and third lines.

**Fig. 2.** Species and functional group frequency of occurrence distributions. Results for grassland (A,D), savanna (B,E) and savanna-forest (C,F) are in first, second and third lines respectively. The points are decreasing order from left to right on the x-axis. The three most frequent ant species and functional groups are indicated in each habitat type.EGP: epigaeic generalist predator; GPC: generalist patrol camponotines; OGD: omnivorous ground-dwellers.