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LAND USE SYSTEMS IN THE AMAZON: IMPACTS ON ANT BIODIVERSITY

Thesis presented in the Federal University of Viçosa as requirement of the Entomology Graduate Program for the achievement of the title of *Doctor Scientiae.*

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Important Notes

This thesis is comprised of four chapters, each representing papers submitted or soon to be submitted to different journals. Individual chapters are composed in accordance with the structure and reference rules requested by the specific journal. They are as follows:

General Introduction: accord of Conservation Biology journal rules.

Chapter I: “Biodiversity declines following the expansion of soybean and cattle ranching in the Amazon” accord of Nature journal rules.

Chapter II: “Soybean Expansion and the Loss of Biodiversity in the Amazon”, accord of the Proceedings of the National Academy of Sciences (PNAS) journal rules.

Chapter III: “The Impact of Forest Plantations of Native and Exotic Species on Ant Biodiversity in the Amazon Region”, accord of Conservation Biology journal rules.

Chapter IV: “Effect of Organic and Conventional Orange Orchard Management on Ant Diversity in the Amazon”, accord of Basic and Applied Ecology journal rules.

General Conclusion: accord of Conservation Biology journal rules.

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Resumo

DOS SANTOS, Iracenir Andrade, D. Sc., Universidade Federal de Viçosa, setembro de 2010. **Sistemas de uso da terra na Amazônia: Impactos sobre a biodiversidade de formigas**. Orientador: Evaldo Ferreira Vilela. Co-Orientadores: Ivette Perfecto, Flávia Maria da Silva Carmo e Simon Loke Elliot.

As florestas tropicais cobrem menos de dois por cento da superfície da terra, mas abrigam cerca de 50% das espécies terrestres do planeta. A região Amazônica é o maior reservatório de biodiversidade no mundo com mais de 40.000 espécies de plantas vasculares descritas, mais da metade das espécies animais descritas no mundo, e a maior bacia hidrográfica com uma extraordinária alta diversidade de organismos de água doce. Porém, essa grande diversidade está altamente ameaçada pelos desmatamentos. A agricultura e a pecuária são as causas primárias do desmatamento nessa região, com a pecuária contando com 65 a 70%, a agricultura em pequena escala responsável por 20-25% e a agricultura em larga escala com outros 5-10%. Nessa região existem muitos tipos de sistemas de uso da terra e a maioria não tem nenhum plano de manejo sustentável. Como esses diferentes sistemas (agricultura, silvicultura e pecuária) são estabelecidos em areas fragmentadas ou mesmo em sistemas naturais como o *Cerrado*, é importante entender a estrutura e a dinâmica das comunidades (animais e plantas) que colonizam esses agroecossistemas. Esse conhecimento nos permitirá medir os impactos sobre a biodiversidade em vários sistemas de produção, bem como o balanço e o potencial sinergético relacionado a objetivos de producão e conserservação. Estudos de biodiversidade, frequentemente, tem usado uma variedade de grupos taxonômicos como indicadores de impactos ambientais. E as formigas são frequentemtemente, usadas como indicadores de biodiversidade e impactos ambientais, em ecossistemas tropicais porque são sensíveis a distúrbios de habitats, tem complexas interações ecológicas com muitos organismos bióticos em ecossistemas terrestres e serem, relativamente, de fácil amostragem e identificação. Nesse estudo foi comparado a diversidade de formigas em uma variedade de agroecossistemas, na região Amazônica brasileira com o objetivo de identificar sistemas que pode ser estimulados e sistemas que devem ser evitados devido aos impactos causados sobre a biodiversidade. Esse trabalho também teve como objetivo entender quais são as variáveis ambientais que podem manter alta biodiversidade entre e dentro de cada tipo de agroecossistema. Para direcionar essas questões foram usados sete agroecossistemas no Estado do Pará, Amazônia Brasileira. Além dos sistemas de uso da terra, foram coletadas um conjunto de variáveis ambientais dentro de cada agroecossistema. No geral, esse estudo mostrou que a serapilheira e a diversidade florística foram diferentes entre todos os agroecossistemas e foram altamente correlacionados com a riqueza de espécies de formigas, sugerindo que essas variáveis desempenham importante papel para a conservação da bioiversidae de formigas dentro das matrizes agrícolas e de pecuária encontradas na região Amazônica. Esse estudo também mostrou que sistemas agroflorestais, plantações de laranjas sob manejo organico e tradicional e florestas plantadas com espécies nativas (*Paricá*) foram os melhores sistemas de uso para conservação da biodiversidade de formigas. Enquanto, plantações de soja, sistemas de pastagens para pecuária e florestas plantadas com espécie exótica (teca) foram os sistemas de uso da terra mais hostís a biodiversdidade de formigas. Também foi evidente que os sistemas que sustentam elevados níveis de biodiversidade são aqueles que tem components arbóreos (sistemas agroflorestais, plantações de laranjas e florestas plantadas com espécies nativas), embora nem todos os

sistemas baseados em árvores foram benéficos para a biodiversidade de formigas, como demonstrado pela baixa diversidade de formigas encontrada em plantações de teca. No entanto, a presença de árvores parece contribuir para a biodiversidade. Portanto, nos processos de tomada de decisão sobre implantação de novos sistemas de uso da terra na Amazônia, é importante considerar sistemas que incorporem árvores, tais como sistemas agroflorestais. Além disso, mais pesquisas devem conduzidas para aumentar e diversificar os agroecossistemas que podem prover hábitas à diferentes espécies e aumentar a biodiversidade, que também possam favorecer processos migratórios entre fragmentos florestais, e assegurar outros serviços ambientais, tais como sequestro de carbono, serviços de polinização, e ao mesmo tempo melhorar a qualidade de vida das populaçõoes locais e regionais.

Abstract

DOS SANTOS, Iracenir Andrade, D. Sc., Universidade Federal de Viçosa, September, 2010. **Land use systems in the Amazon: Impacts on ant biodiversity**. Advisor: Evaldo Ferreira Vilela. Co-Advisors: Ivette Perfecto, Flávia Maria da Silva Carmo and Simon Loke Elliot.

Tropical forests cover less than two percent of the surface of the Earth but they host an estimated 50 percent of terrestrial species on the planet. The Amazon region, in particular, is the largest reservoir of biodiversity in the world, home to more than 40,000 described vascular plant species, more than half of the described animal species in the world and major river basins that contain an extraordinarily high diversity of fresh water organisms. The great diversity of the Amazon is highly threatened by deforestation. Agriculture and cattle ranching are the primary causes of deforestation in the region, with cattle ranching accounting for 65 to 70% of forest clearings, small-scale agriculture for 20-25% and large-scale agriculture for another 5-10%. There are many different land use systems in the Amazon region and most of them do not have any sustainable management planning. As these different systems -- agriculture, forestry and livestock -- get establish and fragment the forest and other natural ecosystems like the *cerrado*, it is important to understand the structure and dynamics of the biodiversity that colonize these land-use systems. That knowledge will allow us to measure the impacts on biodiversity of the various production systems as well as the tradeoffs and potential synergies regarding production and conservation objectives. Studies of biodiversity have often used a variety of taxonomic groups as indicators. Ants are often used as bioindicators of impacts on biodiversity in the tropics ecosystems because they are sensitive to anthropogenic habitat disturbances, they have complex ecological interactions with many other biotic organisms in terrestrial ecosystems and they are relatively easy to sample and identify. In this study we compare ant diversity in a variety of land use systems in the Amazon region of Brazil with the aim at identifying systems that should be promoted and those that should be avoided based on their impact on ant biodiversity. We also want to understand what are the environmental variables that support high ant biodiversity among and within each land use system. To address these questions we used seven land use systems that are found in the State of Pará in the Brazilian Amazon and that have not received much attention from the scientific community, and we sample a set of environmental variables within each system. Overall this study found that leaf litter biomass and floristic diversity of the herbaceous layer were different among all land use systems and were highly correlated with ant species richness, suggesting that they play an important role for the conservation of biodiversity of ants within the agricultural matrices found in the Amazon region. This study also found that agroforestry systems, orange orchard under organic and conventional management and native forest tree plantations (*paricá*) were the best land use systems (of those examined) for the conservation of ant biodiversity in region. While soybean plantations, cattle pastures, and exotic forest tree plantations (teak), were found to be the worst for ant biodiversity. From this study it is also evident that those systems that support the higher ant diversity are those that include trees (agroforestry systems, orange orchards and *paricá* plantations), although not all tree-base systems were beneficial for biodiversity, as demonstrated by the extremely low ant diversity found in teak plantations. Nonetheless, the presence of trees seems to contribute to biodiversity. Therefore, when making decisions regarding land use systems in the Amazon, it is important to consider systems that

incorporate trees, such as diverse agroforestry systems. Furthermore, more research should be conducted on diversifying these systems to provide habitat for biodiversity, promote migration between forest fragments, ensure other ecosystem services, such as carbon sequestration and pollination services, and at the same time contribute to improve the livelihood of the people of the Amazon region.

GENERAL INTRODUCTION

Tropical forests cover less than two percent of the surface of the Earth but they host an estimated 50 percent of terrestrial species on the planet (Feeley & Silman 2009). The Amazon region, in particular, is the largest reservoir of biodiversity in the world, home to more than 40,000 described vascular plant species (Feeley & Silman 2009), more than half of the described animal species in the world (Capobianco et al. 2001; Dirzo & Raven 2003), and major river basins that contain an extraordinarily high diversity of fresh water organisms.

The great diversity of the Amazon is highly threatened by deforestation. Although the rate of deforestation in the Amazon region has been declining since 2004, in 2009 it was reported that more than 7,008 square kilometers were deforested, and between 2000 and 2008 more than 250,000 square kilometers of rainforest were cleared in the Brazilian Amazon (INPE 2010). Agriculture and cattle ranching are the primary causes of deforestation in the region, with cattle ranching accounting for 65 to 70% of forest clearings, small-scale agriculture for 20-25% and large-scale agriculture for another 5-10% (Foley et al. 2007; Gibbs et al. 2008; PRODES/INPA 2010).

With the increase in soybean price in the international market and the introduction of a new soybean variety developed by Brazilian scientists to grow well in the Amazon region, soybean is becoming one of the main causes of deforestation in the region. However, the direct clearing of forest for the establishment of soybean plantations affects only a small area of the Amazon, and the main impact of soybean expansion is indirectly by consuming already deforested land and pushing ranchers and slash-and-burn agriculturalists further into the agricultural frontier (Laurance et al. 2004).

In addition to the dynamic changes that are leading to the expansion of the agricultural frontier in the Amazon, another threat to biodiversity in the region is the unsustainable management of agricultural and forestry systems within the areas that have been cleared for decades. There are many different land use systems in the Amazon region and most of them do not have any sustainable management planning. Over the last decade, the critical role of the agricultural matrix in conserving biodiversity has been recognized by the scientific community (Donald & Evans 2006; Harvey et al. 2008; Hughes et al. 2000; Kumawat et al. 2010; Moorhead et al. 2010; Perfecto & Vandermeer 2002, 2008; Perfecto et al. 2009; Ricketts 2004; Scherr & McNeely 2008; Vandermeer & Perfecto 2007a). In particular, it has been demonstrated that in fragmented landscapes, the agricultural matrix plays a critical role in the conservation of biodiversity by facilitating or hindering migration of organisms to forest fragment (Perfecto & Vandermeer 2008, 2010). Diverse agroforestry systems that do not use toxic chemicals and resemble forests allow organisms to move through the agricultural matrix and migrate to forest fragments, while intensive annual monocultures that use toxic chemicals, eliminate ground cover and extend over very large areas, prevent forest organisms from migrating from one forest fragment to another. Without the necessary gene flow and immigration, forest organisms are bound to local extinctions and with the recolonization potential minimized, those local extinctions will eventually become regional or even global (Perfecto et al. 2009).

At a very large scale, the agricultural matrix in the Amazon region consists of a mosaic of many different land-use systems. As these different systems -- agriculture, forestry and livestock -- get establish and fragment the forest and other natural ecosystems like the *cerrado*, it is important to understand the structure and dynamics of the biodiversity that colonize these land-use systems. That knowledge will allow us to measure the impacts on biodiversity of the various production systems as well as the tradeoffs and potential synergies regarding production and conservation objectives. By understanding how these systems impact biodiversity, and what environmental variables contribute the most to this diversity, we will be able to develop more biodiversity-friendly systems for the agricultural matrix within the Amazon region. Since biodiversity has been linked to a variety of ecosystem services (Balvanera et al. 2006; Swift et al. 2004; Tscharntke et al. 2005), as well as to sustainability and productivity of managed ecosystems (Balirwa et al. 2003; Brussaard et al. 2007; Finegan et al. 1998; Primavesi 2006; Scherr & McNeely 2008; Tilman 1999; Tilman et al. 2002), promoting biodiverse systems will not only conserve biodiversity within the agricultural matrix, but it can also contribute to the productivity and sustainability of production systems within the Amazon region.

Studies of biodiversity have often used a variety of taxonomic groups as indicators (Lawton et al. 1998; Lindenmayer et al. 2000; Schulze et al. 2004). Although no single taxonomic group has been found to be a good surrogate for all other groups, most groups are affected in similar ways by habitat modification and follow similar patterns of species diversity (Lawton et al. 1998; Schulze et al. 2004). Ants are often used as indicators of biodiversity in the tropics ecosystems because they are sensitive to anthropogenic habitat disturbances, they have complex ecological interactions with many other biotic organisms in terrestrial ecosystems and they are relatively easy to sample and identify (Andersen et al. 2004; Andersen et al. 2002; Armbrecht et al. 2005; Floren & Linsenmair 2005; Roth et al. 1994).

In the Brazilian Amazon, ants are extremely diverse and abundant. According to Wilson (1971) there are more ant species in a square kilometer of Brazilian entire animal biomass of the *terra firme* forest in the Amazon is composed of ants and termites (Holldobler & Wilson 1990; Wilson 1971). However, the importance of ants is more related to the diversity and strength of the interactions they have with other organisms and their functions as ecosystem engineers (Folgarait 1998; Holldobler & Wilson 1990; Vandermeer & Perfecto 2007b).

In this study we compare ant diversity in a variety of land use systems in the Amazon region of Brazil with the aim at identifying systems that should be promoted and those that should be avoided based on their impact on ant biodiversity. We also want to understand what are the environmental variables that support high ant biodiversity among and within each land use system. To address these questions we used seven land use systems that are found in the State of Pará in the Brazilian Amazon and that have not received much attention from the scientific community, and we sample a set of environmental variables within each system.

The first paper focuses on two of the fastest-growing and most controversial land use systems in the Brazilian Amazon today, soybean plantations and cattle pastures. Since soybean plantations tend to be established in cattle ranches, which are then displaced to the agricultural frontier, we sampled ants in both of these systems. Furthermore, since agroforestry systems (AFS), have been suggested as an alternative and appropriate form of agriculture for reducing the environmental impacts of agriculture in the region (Anderson 1990; Finegan et al. 1998; Fujisaka & White 1998; Nicholaides et al. 1985) we also sample ants in AFS. Finally, using data from forest fragments that was collected in 2009 using the same methodology used in the current study, we compare ant species richness of all three land use systems with that in forest fragments in the same region. In addition to comparing ant species richness in all three production systems and forest fragments, we developed a model using Conversion System Rate Model (CSRM) to simulate ant species richness under various scenarios of land use. The first scenario examined the transformation of the landscape from cattle pasture to soybean; the second examined the transformation from forest to cattle pasture; and the last two, examined the transformation of both soybean and cattle pasture to agroforestry systems.

The second paper compares estimated species richness, abundance and composition of the ant community in the same three land use systems, soybean plantations, cattle pastures and agroforestry systems, and examined in more detail the environmental variables that contribute to ant species richness in each of them. Using predictive bayesian models we predict ant species richness based on the various environmental variables sampled.

Tree plantations are increasingly favored land use systems in the Amazon. The National Forest Program has as one of its key objectives the reforestation of degraded lands with tree plantations, with plans to cultivate 22 million hectares of tree plantations on private and abandoned agricultural lands (PNF 2000). Carbon sequestration projects, including the recent Reducing Emissions from Deforestation and Forest Degradation (REDD) program, as well as a growing interest in biofuels and timber, will likely result in an expansion of tree plantations in the world (Pacala & Socolow 2004), as well as in the Amazon region (Yu 2004). In the third paper we examined ant diversity in two types of timber plantations in the Amazon region. We compared ant species diversity under teak

(*Tectona grandis*), an exotic species often used in timber plantations throughout the tropics (FAO 2009), and especially in the Amazon region (ABRAF 2009), and under paricá (*Schizolobium amazonicum*), one of the most common native trees planted for timber in the region. This study also examines which environmental variable best explained ant biodiversity in native- and exotic-planted forests and discuss the tradeoffs associated with these two types of plantations.

Finally, since organic systems are increasing worldwide due to a higher demand for these products (IFOAM 2010), in the last paper evaluate ant species richness and composition in orange orchards under organic and conventional management. In this paper we also investigate what environmental variables contribute the most to the maintenance of ant diversity in these systems.

The last chapter of this thesis summarizes the main findings and conclusions of the studies and discusses alternatives for enhancing the agricultural matrix in the Amazon region emphasizing systems that create synergies between conservation and livelihood objectives of the people of the Amazonia.

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CHAPTER I

BIODIVERSITY DECLINES FOLLOWING THE EXPANSION OF SOYBEAN AND CATTLE RANCHING IN THE AMAZON

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Abstract

It is familiar knowledge that the expansion of pastures and soybeans in the Amazon contributes to biodiversity loss. Yet a quantitative accounting of that loss has focused on hectares deforested with less regard for the actual impacts on biodiversity over the entire landscape. Here we use ants as an indicator group to study species diversity in forests, pastures, soybean plantations and agroforestry systems in the Amazon region. We also developed projections of species richness under various scenarios with different proportions of these agricultural systems at the landscape level. We find that ant species richness in pasture and soybean is only 25% and 20% of that in primary forest, while agroforestry systems contain 83% of the species richness of the forest. Furthermore, projection models based on species richness estimates indicate that agricultural landscapes dominated by either pasture or soybean will harbour extremely low diversity as compared to landscapes dominated by agroforestry systems.

The loss of biodiversity is one of the major environmental challenges facing humanity, with deforestation of tropical rain forests receiving the bulk of attention. In the Brazilian Amazon, conversion to agriculture is frequently the proximate cause of deforestation. Brazil is the world’s largest exporter of beef and the second largest exporter of soybeans,[1] contributing significantly to the economic performance of this country,[1,2]. The recent growth of both of these sectors has contributed to the deforestation crisis in Brazil.[2,3] Concern has been raised as to the effect that massive soybean expansion in the Amazon might have on this especially biodiversity rich area of the world.[3-5] Partly in response to this concern in 2006 a two-year moratorium on converting forest to soybeans in the Amazon basin was announced by the Brazilian Association of Vegetable Oil Producers and the Brazilian National Grains Exporters’ Association (ABIOVE/ANEC), the two major Brazilian soybean traders. In 2008 it was extended for one more year with the additional cooperation of the Brazilian Ministry of the Environment, and since has had one-year renewals in July 2009 and 2010.[6] As a response to this moratorium, soybean producers began expanding their production to former pasture lands.[7] Not surprisingly, the continued demand for beef products resulted in further expansion of pastures into the forest.[5-9] Although deforestation rates in the Amazon have been declining since 2004, in 2009 more than 7,000 square kilometers of forest were cleared.[10] In other words, the socio-political dynamics of the situation presently are for forest to be converted to pasture, which is subsequently converted to soybeans, all in conformity with the moratorium against expanding soybean production directly into forested areas.

Another type of agricultural system, practice primarily by smallholders in the Amazon, is agroforestry, which consists of the use of timber or fruit trees along with crops or pastures. Agroforestry systems have long been a traditional form of production in the humid tropics,[11] including many areas of the Amazon basin, and have been proposed as productive alternatives to soybean and pastures.[12-17]

Although the environmental impacts of converting large tracks of primary forests to pastures and soybean are evident and well documented in terms of the loss of habitat,[18] there are no quantitative studies of the biodiversity value of soybean, nor any comparisons of biodiversity with the habitats they replace, particularly pastures and agroforestry systems.[19] Given the current rate of conversion of forest to pasture and pasture to soybean in some areas of the Amazon basin,[20] it is imperative that some quantitative measure of the biodiversity conservation potential of these land use systems be made, and compared to the original forest.

To this end we used ants as an indicator group (see supplementary information) and systematically sampled soybean plantations, pastures and agroforestry systems in the state of Para in the Amazon region of Brazil. We collected ants using pitfall traps in the three agroecosytems from January to April, 2008. To compare biodiversity within these land use systems with primary forests, we used pitfall trap data collected in August, 2009. We compared ant species richness using the expected species accumulation curves based on observed species richness (Mao Tau adjustment) and Chao1 estimator in EstimateS,[21] and then employed a Conversion System Rate Model (CSRM) based on a re-sampling analysis to predict ant species richness under various scenarios of landscapes composed of different proportions of pasture, soybean and agroforestry systems. We used the same model to predict the number of ant genera under three scenarios of primary forest conversion – forest to pasture, forest to soybean and forest to agroforestry (see supplementary information). Further exploration of the effects of distance from the nearest forest fragment was conducted using predictive Bayesian models. For detailed information on methods and analyses see supplementary information and supplementary Table S1.

Soybean and pasture sites had significantly lower estimated species richness than that found in agroforestry or primary forest sites (Fig. 1). Agroforestry systems contained 83% of the estimated ant species richness of the primary forest, a difference that was not statistically significant (Fig. 1). Pastures had only 25% and soybeans 20% of the estimated species richness of forests (statistically significant in both cases). Furthermore, pastures had 31% and soybean 24% of the estimated species richness of agroforestry systems, and the pattern of species accumulation with increasing sampling intensity suggests that our methods were actually more efficient in soybean fields and pastures than in agroforestry systems and primary forests (see supplementary information, Fig. S1). Consequently, the estimates of species decline with conversion to soybean or pasture are conservative. Finally there were 41 ant species found in agroforestry sites which were not found in pasture or soybean sites, compared to just four species unique to pasture and five to soybean, the latter being species well-known from human-influenced environments (see supplementary information for ant species list, Table S2).

Application of the conversion system rate model provides us with an estimate of ant species richness within the agroecosystem sector, based on the fraction of land devoted to each agroecosystem (Fig. 2). With either a soybean or pasture-dominated landscape the ant biodiversity would decline dramatically (Fig. 2a and b). As pasture is replaced by soybean in our re-sampling simulations, as is currently happening in the Amazon region, there is no decrease in ant species richness because the two land use systems have a similar number of species (Fig. 2c). It is not until more than 90 percent of the landscape is converted from pasture to soybean that there is a decline in ant species richness. In fact because their ant communities are not completely overlapping, a landscape partially composed of soybean and pastures contains more species than a landscape composed entirely of either one. Further analysis (see supplementary information, Fig. S2) demonstrates that although conversion from forest to agroforestry systems results in a small loss of biodiversity, our model estimates that the loss will be far more dramatic when primary forest is converted to pasture, and even more so when the forest is converted to soybean. Looking at a matrix landscape composed of the three land use systems (Fig. 2d) as the proportional mix increases in either soybean or pasture – particularly soybean – dramatic declines in species richness are to be expected and, strictly from the point of view of biodiversity conservation, maximizing the area devoted to agroforestry is the best way to conserve biodiversity in the agricultural landscape.

In addition to the direct conservation value of particular agroecosystems, recent literature has emphasized the importance of the agroecosystem as a matrix in which an overall landscape, including natural forest fragments, would be biodiversity-friendly.[22,23] In short, a “high quality” or “permeable” matrix is one in which migration among forest fragments is encouraged.[24] Parsing our data to examine sampling sites near to forest fragments versus far from forest fragments (see supplementary information) we find a significant effect (Fig. 3). Biodiversity in the agroforestry system is maintained at a high level even far from forest fragments, suggesting that it might represent a high quality matrix, whereas the biodiversity in soybean fields is reduced in samples far from forest fragments. Indeed, in sites far from the forest, ant species richness in agroforestry sites show a slight, but statistically significant increase in ant species richness, suggesting that in highly deforested areas, agroforestry systems could represent a refuge for biodiversity.

We conclude that the expansion of soybeans, whether directly through forest clearing or indirectly through pasture conversion, results in a significant decline in biodiversity. The promotion of agroforestry systems in the Amazon region would have a significant positive effect on biodiversity conservation, either as a habitat for biodiversity, in and of itself, or as a high quality matrix that can increase permeability for forest species.

Supplementary Information is linked to the online version of the paper at www.nature.com/nature.

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Author Contributions I.A.D.S. was involved in study design, collected and analyzed the data, performed the identification of the ants, and wrote the paper; D.N.A. wrote the paper and developed the CSRM model; I.I. developed the EVPM Bayesian model and provided significant intellectual input to the manuscript; J.Q. collected and identified the ants for the primary forest sites; J.V. and I.P. provided significant intellectual input to the manuscript; E.F.V and O.R.K discussed the results and commented on the manuscript.

Author information The ants collected in this study have been deposited at Regional Entomology Museum in Federal University of Viçosa. The authors declare no competing financial interest.

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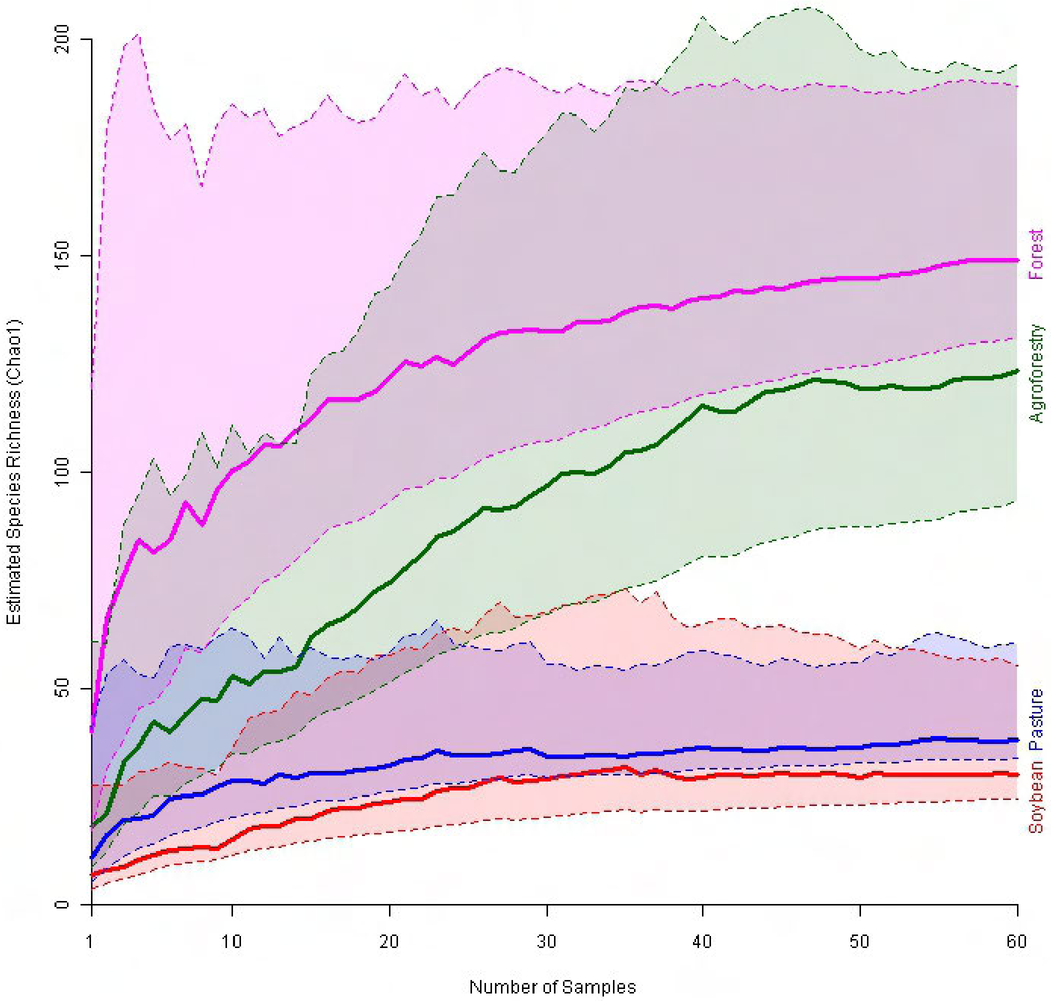
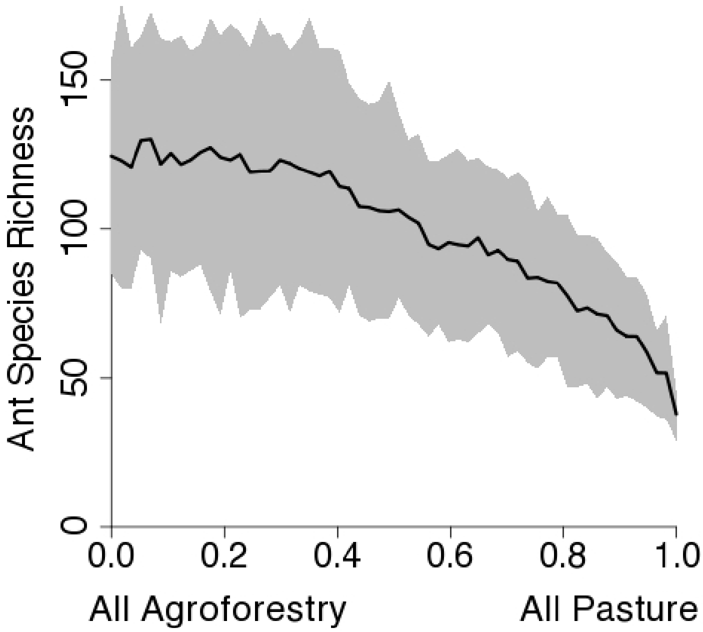
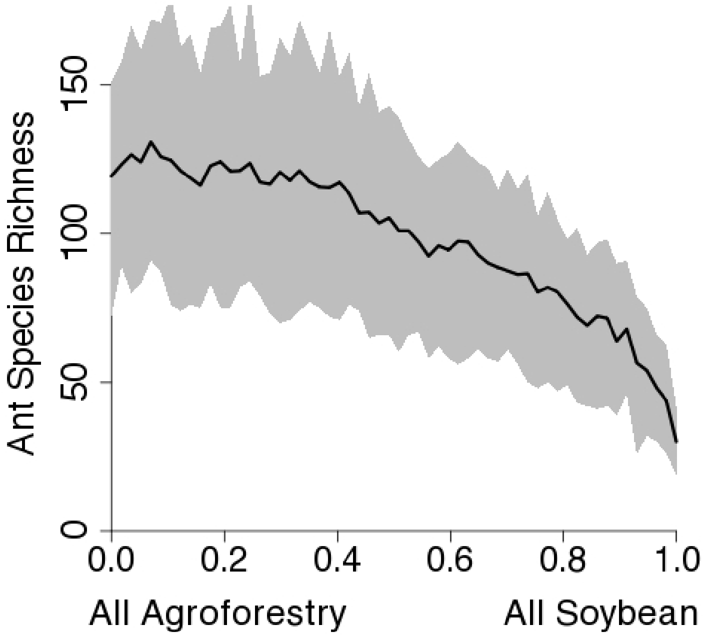


Figure 1. Estimated species accumulation curve (Chao1). Solid lines are the mean estimated number of species at specific number of samples, dotted lines embracing constant colour track the 95 percent confidence limits on the estimates. The pink line is forest; green line is agroforestry; blue line is pasture; and red line is soybean. Note the broad overlap in 95 percent confidence limits for forest and agroforestry, and distinct separation of those limits for pasture and soybean at sample sizes greater than 35 pitfall traps.

A


Ant Species Richness
0
50
100
150
0.0
0.2
0.4
0.6
0.8
1.0
All Agroforestry
All Pasture

B


Ant Species Richness
0
50
100
150
0.0
0.2
0.4
0.6
0.8
1.0
All Agroforestry
All Soybean

C

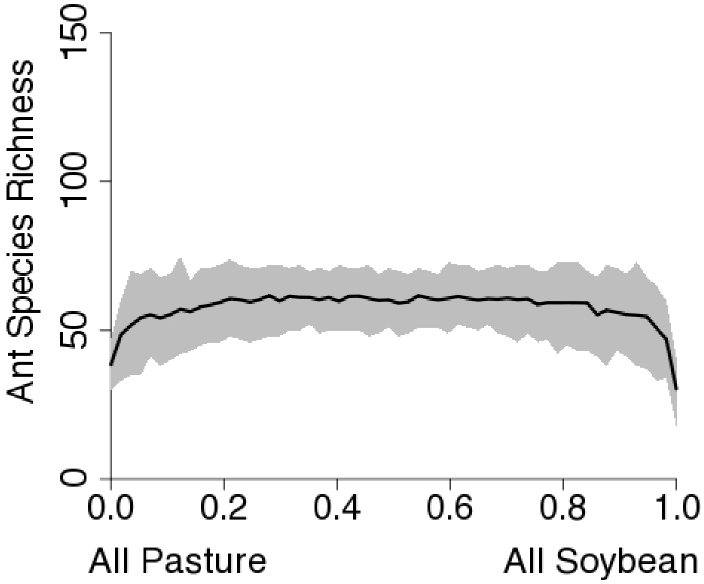


Figure 2. Predictions from the Conversion System Rate Model for ant species richness. A) Mean number of species with 95 percent confidence intervals for agroforestry to pasture conversion. The x-axis is the proportion of the landscape converted to pasture. B) Conversion from agroforestry to soybean. C) Conversion from pasture to soybean.

D

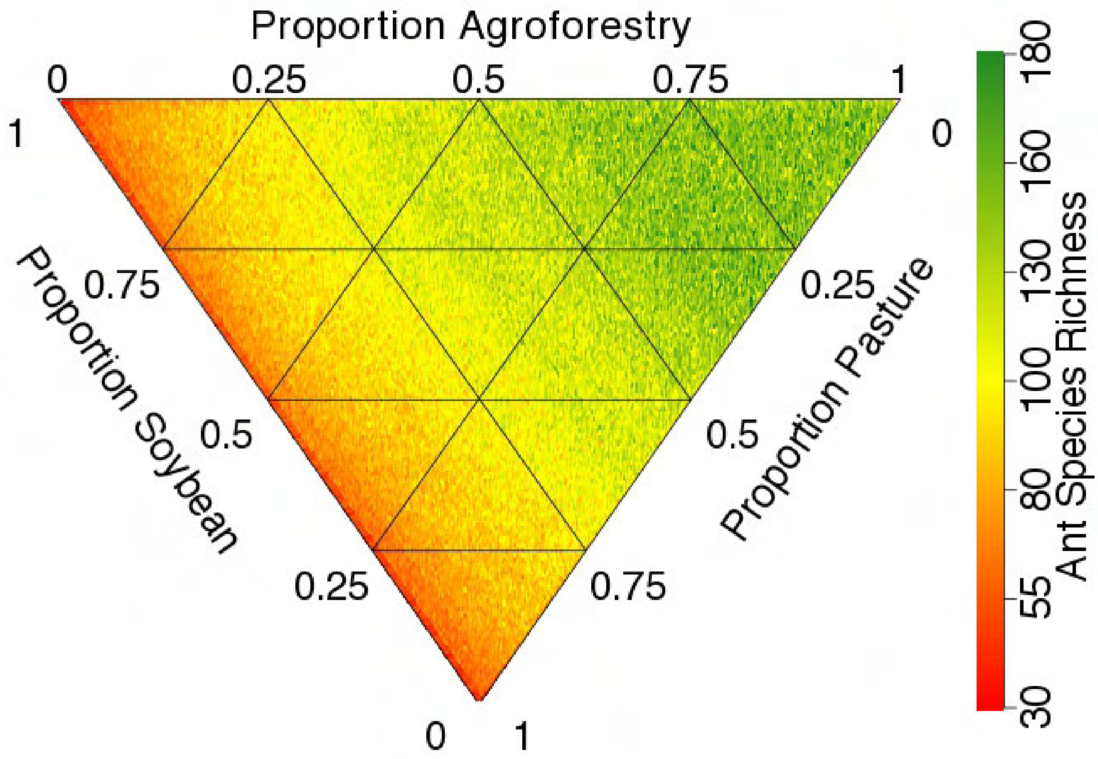


Figure 2. Predictions from the Conversion System Rate Model for ant species richness. D) Predictions from the Conversion System Rate Model for a mixture of the three agroecosystems. The colour represents the mean number of species in 25 runs.

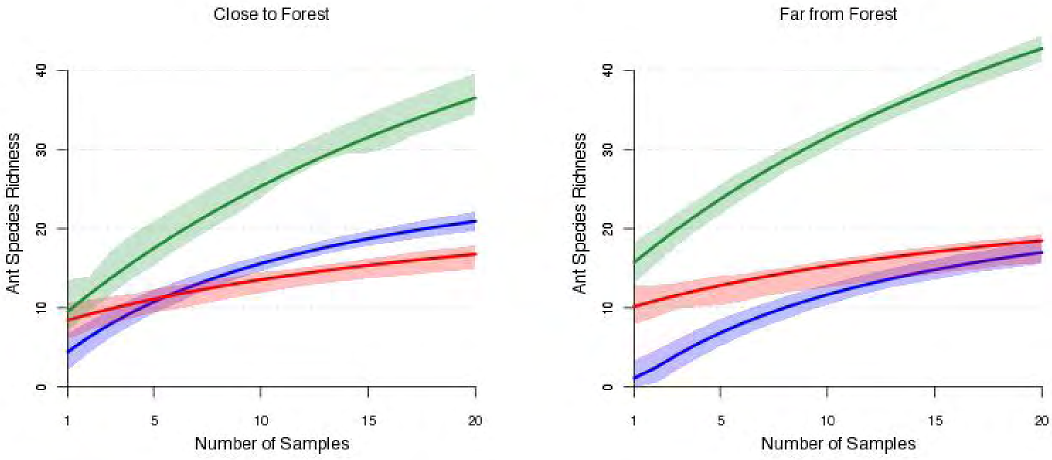


Figure 3. Species accumulation curves from a predictive Bayesian model (described in detail in the supplementary information) for the three agroecosystems close to and far from a forest fragment. The green line is agroforestry; blue line is pasture; and red line is soybean.

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SUPPLEMENTARY INFORMATION

Methods

Study Site

The three agroecosystems (soybean, pastures and agroforestry) were sampled between January and April, 2008 in Paragominas (soybean plantations- around 03°04'50"S; 47°22'42"W), Igarapé-Açú (Pasture plantations- around 01°04'36.6"S; 47°36'48"W) and Igarapé-Açú and Inhangapí (agroforestry systems, around 01°02'39.7”S; 47°35'39.9"W). We used three replicates for each system. The district of Paragominas is the main soybean production area in the State of Pará and, in the last nine years, has been the region with the second highest rate of deforestation, second only to São Félix do Xingu.[1] The other two districts in the study have substantial areas devoted to agroforestry and pasture. The forest samples were part of a separate study and were collected in August 2009 in primary forest sites within the National Forest of Tapirapé-Aquiri, Serra de Carajas, Pará State (site a- around 05°49'24.6"S; 50°31'50.1"W; site b- around 05°46'22"S; 50°29'54.1"W; and site c- around 05°46'19.8"S; 50°33'53"W).

Ant Sampling

For the three types of agroecosystems we collected ants using pitfall traps (18 cm diameter and 10.5 cm height). In each replicate we installed 20 traps 15 m apart, arranged in four rows of five pitfalls. We left the traps open for 48 hours. Specimens were sorted, mounted and identified to genus and species or morphospecies based on literature[2,3] and on comparisons with specimens of the Formicidae Collection of the Museum of Zoology of the University of São Paulo. The forest ants were also identified to genus and morphospecies but since another research team conducted the study, the morphospecies classification does not necessarily correspond to precisely the same species as in the three agroecosystems and therefore overlap of species between forest and the other three sites could not be estimated.

Measuring distance from the nearest forest fragment

For each pitfall trap the distance from the forest was determined by measuring the distance between the center of the pitfall trap and the closest forest using a 200m measuring tape. When distances were large -- more than 200m and the distance to the nearest forest fragment was ambiguous (i.e. there were several forest fragments at similar distances) -- we used a Topcon Pulse Total Station GPT 3000 to measure various distances and determine the shortest distance. Once that point was selected we measured the distance with a 200m measuring tape. This was done to avoid errors generated by the Topcon Pulse Total Station GPT 3000 under conditions of high humidity.

Ants as an Indicator Group

Studies of biodiversity have often used a variety of taxonomic groups as indicators.[4-6] Although no single taxonomic group has been found to be a good surrogate for all other groups, most groups are affected in similar ways by habitat modification and follow similar patterns of species diversity.[4,6] Ants are often used as indicators of biodiversity in tropical ecosystems because they are sensitive to anthropogenic habitat disturbances, they have complex ecological interactions with many other biotic organisms in terrestrial ecosystems and they are relatively easy to sample and identify.[7-11]

In the Brazilian Amazon, ants are extremely diverse and abundant. One third of the entire animal biomass of the *terra firme* forest in the Amazon is composed of ants and termites.[3] However, the importance of ants is more related to the diversity and strength of the interactions they have with other organisms and their functions as ecosystem engineers.[3,12,13]

Prediction Models

Conversion System Rate Model (CSRM)

We conducted a resampling analysis to predict ant species diversity in landscapes composed of different fractions of the three habitat types. For each habitat type we calculated the number of samples it took for the Chao1 species accumulation curve to asymptote (Figure 1). In our hypothetical landscapes we sampled each habitat by taking a number of samples equal to a fraction of that asymptote-sample number. We used Chao1 estimates to count the number of species found within the pooled samples from across the fractional habitats, making sure not to double count the overlapped species, which Chao1 also estimates. For each habitat-fraction combination we repeated the resampling 100 times. Figure 2a-c shows the results for hypothetical habitats composed of two of the three habitat types with the 95 percent confident intervals for the 100 runs, and Figure 2d for those composed of a combination of all three. To compare these three agroecological habitat types to the forest we conducted the above analysis at the genus level since we could not verify the congruency among the morphospecies in the forest sites and the agroecosystems (Figure S2). In recent years it has been demonstrated that analyses at the genus level represent taxonomic sufficiency for Neotropical ant assemblages[14].

Environmental Variables Prediction Model (EVPM)-Distance from the Forest

In this study we collected data on many environmental variables, however, here we report the results for only distance from the forests because it was our only landscape level variable and the most relevant to the data presented here. The effects of vegetation and soil variables on ant species richness are reported elsewhere (Dos Santos et al., in preparation). To estimate ant species richness as a function of the distance from the forest we developed a Bayesian model.[15,16] The Bayesian framework allowed us to estimate different responses at each of the three agroecosystems while still having an overall response to the variability sampled. The Bayesian method was also useful to deal with the variability found among the environmental variables (see below).[17]

Likelihood

From the expected species accumulation curves (Mao Tau) estimates, the number of accumulated species of ants, *S*, found in each system, *s*, at each replicate or farm, *f*, for a certain number of traps, *nt*, is modeled following a normal distribution that constitutes the likelihood of the model:

$$S_{s,f,nt} \sim Normal\left(\mu_{s,f,nt}, \sigma_{nt}\right)$$

The mean, *μ*, is then estimated as a combination of the number of traps sampled, *nt*, which follows a saturation curve, and of the environmental variables measured. We were particularly interested in the effects of distance to the nearest forest fragment[18] on ant diversity, but we are also aware that other environmental variables may affect diversity and consequently included them in the analysis to better evaluate the effects due solely to distance, the final model was then:

$$\begin{aligned}\mu_{s,f,nt} = {} & \frac{\alpha_s nt}{nt + \theta_s} + \beta_1 NP_{s,f} + \beta_2 DP_{s,f} + \beta_3 Dist_{s,f} + \beta_4 Biom_{s,f} \\ & + \beta_5 SD_{s,f} + \beta_6 N_{s,f} + \beta_7 OM_{s,f} + \beta_8 pH_{s,f} + \beta_9 K_{s,f} \\ & + \beta_{10} Na_{s,f} + \beta_{11} Ca_{s,f} + \beta_{12} PH_{s,f}\end{aligned}$$

The full model included the following environmental variables: NP, plant species number; DP, plant density; Biom, leaf-litter biomass; SD, soil depth; N, soil nitrogen content; OM, soil organic matter; pH, soil pH; K, soil potassium; Na, soil sodium; Ca, soil calcium; and PH, plant height. These variables were measured at each ant baiting site. The parameters *α*, estimated for each system as a lognormal distribution (to ensure positive values), determine the expected maximum number of species at each system. The parameters θ represent the half saturation point along the species accumulation curves, i.e., how many traps are needed to sample half of the number of species in the system. Both sets of parameters were estimated at the system level as:

$$\alpha_s \sim logNormal(0,1000)$$

$$\theta_s \sim Uniform(0,100)$$

Fixed effect coefficients, *β*, were estimated from normal distributions with uninformative parameter values:

$$\beta_k \sim Normal(0,10000)$$

These *β* values determine the contribution of each environmental variable. To better compare these coefficients we used standardized values of the environmental variables (standardized value: [variable value – variable mean]/[2 variable standard deviation]).[16]

The precision, τ, (inverse of the variance σ) estimated at each number of traps was modeled from a gamma distribution. We expected larger variances at a low number of traps as they almost cover the full range of number of species observed at each trap, and at high number of traps as the sample sizes decrease (e.g., for 20 traps sample size we only have one observation per farm):

$$\tau_{nt} \sim Gamma(0.01, 0.01)$$

All parameter priors and initial values used in the runs were non-informative. Models were run in OpenBUGS[19] for 100000 iterations to ensure convergence of the parameters. Burn-in iterations were discharged in the estimation of the posterior parameter means.

Restricted Environmental Variables Model

Since many of the environmental variables were strongly correlated we choose three of them that seemed to be representative of all the others: *distance to forest fragment, herbaceous plant density and leaf-littler biomass*. The environmental variables included in the analysis were then considered to be latent variables that were also estimated as part of the model. For each number of pitfall traps, *nt*, at each replicate or farm, *f*, the value of each environmental variable was estimated based on the farm's mean and standard deviation, estimated values were then standardized (see above) for the analysis:

$$env.\mathrm{var}._{f,nt} \sim Normal\left(env.var.mean_f, env.var.variance_f\right)$$

In this paper we show the results only for distance to the nearest forest fragment because this is the only landscape-level variable that we recorded, and the most pertinent to the results reported here.

RESULTS

Species Accumulation Curves

In addition to the species accumulation curves based on the Chao1 estimates, we constructed species accumulation curves (sample-based rarefaction) using the observed (Mao Tau) adjusted estimates with 95% confidence intervals. These curves show significant differences in species richness between the forest sites and the agroforestry sites (Fig. S2). However, as can be seen from Figure 1 in the main text, once Chao1 estimates are calculated, that difference disappears (using 95% CI). As with the Chao1 estimates (Fig. 1 in main text), these curves show that both the soybean and the pasture sites have significantly lower ant species richness than either the forest or the agroforestry sites.

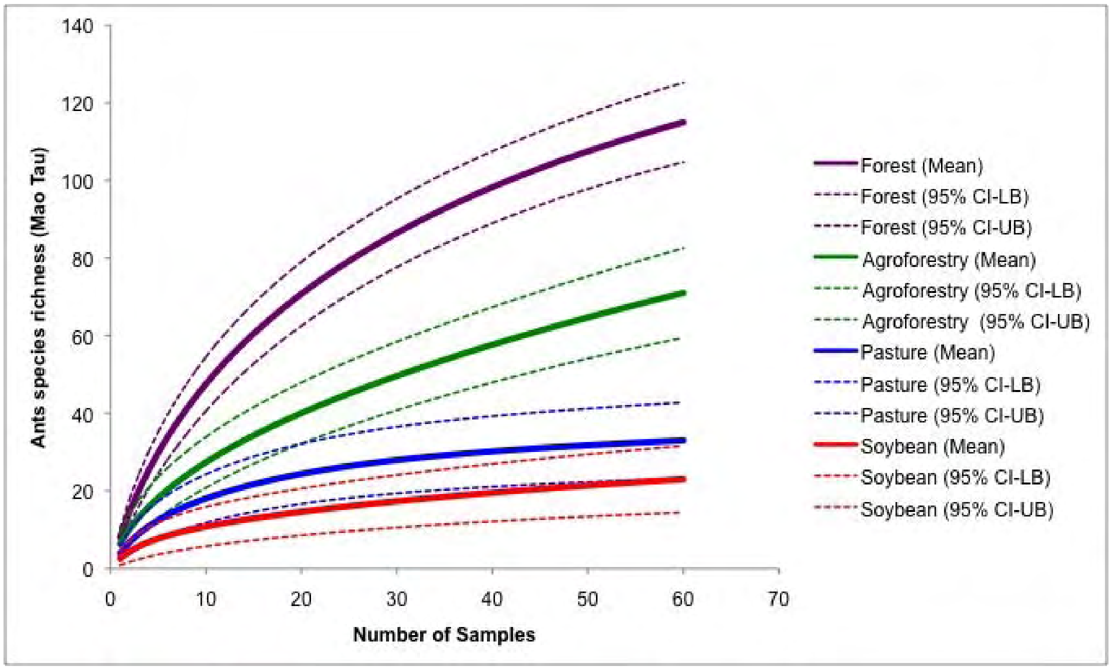
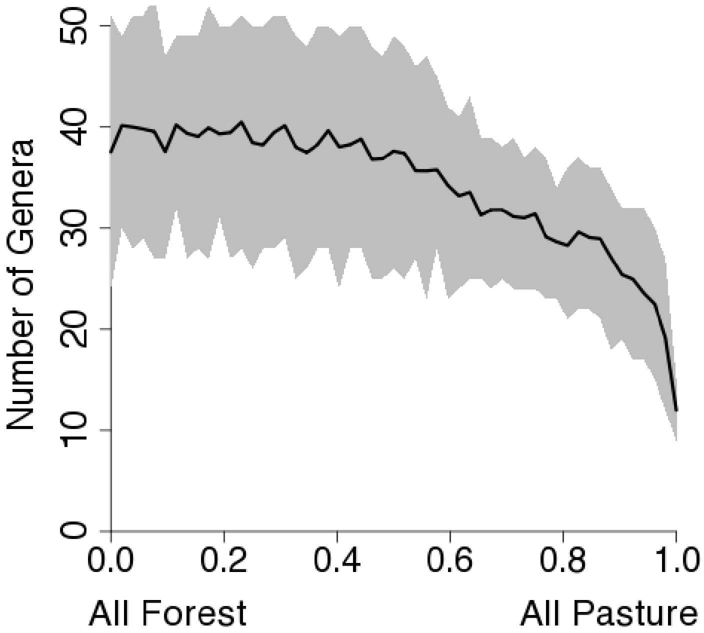


Figure S1. Species accumulation curves for forest, agroforestry, pasture and soybean based on the Mao Tau adjustment, with 95% confidence intervals.

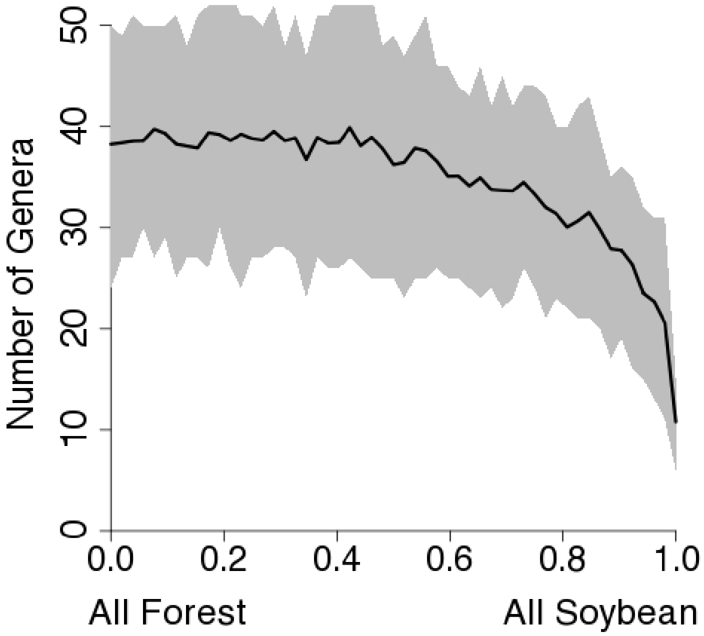
Conversion System Rate Model – Conversion of Forest to Agriculture Using Genus as the Taxonomic Unit of Analysis

The Conversion System Rate Model was run to examine the change in the number of ant genera when primary forest is converted to agriculture (see methods above). The results predict a large loss in ant genera with conversion to pasture and soybean but a smaller decline when the transformation is to agroforestry systems (Figure S2).

A


Number of Genera
0
10
20
30
40
50
0.0
0.2
0.4
0.6
0.8
1.0
All Forest
All Pasture

B


Number of Genera
0
10
20
30
40
50
0.0
0.2
0.4
0.6
0.8
1.0
All Forest
All Soybean

C.

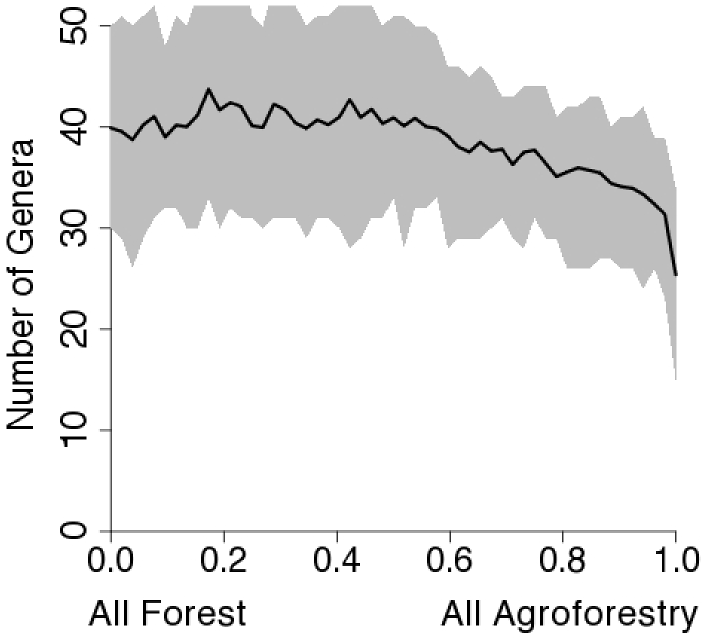


Figure S2. Predictions from the Conversion System Rate Model for ant genus richness. A) Mean number of species with 95% confidence intervals for forest to pasture conversion. The x axis is the proportion of landscape converted to pasture. B) Conversion from forest to soybean. C) Conversion from forest to agroforestry.

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Supplementary Table S1– EVPM Results

Posterior mean +SD (95% credible interval).

Fixed effects coefficients

Variable coefficient	**Mean +SD (95% CI)** Coefficients statistically significant, different from zero, are indicated in bold
Plant Species - NP	-7.25 +0.57 (-8.94,-6.33)
Plant Density - DP	-1.89 +0.6 (-2.91,-0.9)
Distance from the Forest - Dist	0.78 +0.61 (-0.36,1.89)
Leaf Litter Biomass - Biom	10.99+0.92 (9.28,12.13)
Soil Density - SD	-6.39 +0.23 (-6.74,-5.77)
Nitrogen - N	11.44 +0.41 (10.43,12.04)
Organic Matter - OM	24.18 +0.94 (22.58,25.51)
soil pH - pH	-4.35 +0.42 (-5.03, -3.5)
Potassium – K	3.95 +0.94 (20.6, 5.11)
Sodium – Na	3.04 +0.56 (2.01, 4.03)
Calcium – Ca	6.03 +0.17 (5.7,6.38)
Plant High – PH	1.11 +0.73 (0.03,6.38)

Parameters α (maximum number of species) and θ (half saturation) from the saturation curves

Parameter	**Mean +SD (95% CI)**
α pasture	31.2 +0.66 (30.01, 32.27)
α soybean	18.2 +1.04 (20.3,20.72)
α agroforestry system	76.75 +1.64 (73.09,78.2)
θ pasture	13.44 +0.89 (12.5,14.7)
θ soybean	24.31 +2.96 (20.1,26.2)
θ agroforestry system	32.36 +0.98 (30.84, 33.35)

Estimates of the variance

Number of pitfall traps sampled	Mean +SD (95% CI)
1	8.46 +19.4 (4.27, 23.7)
2	5.11 +11.52 (2.5, 14.9)
3	3.71 +7.85 (1.75,11.53)
4	2.76 +5.59 (1.25, 9.19)
5	2.05 +4.09 (0.93, 7)
6	1.48 +2.89 (5.32, 0.67)
7	1.06 +1.96 (3.88,0.45)
8	0.69 +1.18 (0.28, 2.73)
9	0.41 +0.64 (0.16, 1.76)
10	0.20 +0.22 (0.06, 1.1)
11	0.063 +0.03 (0.01, 0.56)
12	0.017 +0.013 (0.002, 0.21)
13	0.0106 +0.011 (0.003, 0.11)
14	0.015 +0.011 (0.004, 0.24)
15	0.05 +0.03 (0.01, 0.48)
16	0.14 +0.16 (0.046, 0.82)
17	0.26 +0.37 (0.1, 1.18)
18	0.4 +0.62 (0.15, 1.73)
19	0.55 +0.94 (0.22, 2.2)
20	0.71 +1.27 (0.3, 2.8)

Supplementary Table S2– List of Ant Species

a) Ants from the three agroecosystems:

Ant species	Past1	Past2	Past3	Soy1	Soy2	Soy3	Agrof1	Agrof2	Agrof3
Dolichoderinae									
Dolichoderus bispinosus								1	
Dolichoderus diversus									1
Dolichoderus quadridenticulatus							1		
Dorymyrmex (px) *jheringi*		1					13	13	9
Dorymyrmex brunneus		6	1			2	4		9
Dorymyrmex spurius		4				1	2	1	4
Ecitoninae									
Labidus coecus							1		
Nomamyrmex esenbeckii					1				
Ectatomminae									
Ectatomma brunneum	12		11				2	16	
Ectatomma tuberculatum							1		
Gnamptogenys rastrata							1		1
Formicinae									
Acropyga palaga	1								
Brachymyrmex pictus							1		
Brachymyrmex sp20							1		
Brachymyrmex sp21							1		
Brachymyrmex sp22							1	1	
Camponotus sp21								2	
Camponotus cingulatus					1				
Camponotus crassus							2	4	
Camponotus renggeri							3	1	
Camponotus sp10									1
Camponotus sp20								1	
Camponotus sp21								1	
Camponotus sp4							1		
Paratrechina fulva	1	5	1					5	1
Paratrechina sp20								1	2
Myrmicinae									
Acromyrmex laticeps nigrosetosus								1	
Mycetarotes paralellus							1		
Oxyepoecus nsp								1	1
Oxyepoecus vezenyii							1		
Pheidole (gr. *fallax)* sp36	2		1			3			3
Pheidole (gr. *flavens)* sp1								1	
Pheidole (gr. *flavens)* sp36	2	2	1						
Pheidole fallax	4	13	4	12	24	18	9	8	15
Pheidole flavens	2			2	1	4	1		
Pheidole sp20						1		1	
Pheidole sp24								1	
Pheidole sp25	4	3	1	8	5	6		3	3
Pheidole sp26	2		3					1	1
Pheidole sp27								2	
Pheidole sp35		1				2			1
Pheidole sp40								2	

Pheidole sp45								1	
Pheidole sp6	1	1	2					1	1
Solenopsis geminata	2	2		4	9	3			
Solenopsis globularia	3	9	2	4		1	3	2	6
Solenopsis invicta	2	2		1	7	10			1
Solenopsis molesta	2	3	2					2	6
Solenopsis saevissima	4	17	18	12	7	1	20	15	16
Solenopsis spXX		2		2	2				4
Strumigenys eggersi	1		3	1				3	
Strumigenys grytava								2	3
Strumigenys louisianae						1			
Wasmannia auropunctata	12		12					5	1
Cardiocondyla nuda	5								
Cephalotes atratus							1		
Cephalotes pusillus							1		1
Crematogaster abstinens	1	1	3			2		2	7
Crematogaster quadriformis	5	7	10		3	7	2	5	7
Crematogaster sp21						1			
Crematogaster sp30			1						
Crematogaster sp31							1		
Cyphomyrmex sp1	2						1		
Cyphomyrmex sp11							1		1
Cyphomyrmex sp2	2		1				1		
Cyphomyrmex sp3	1						2		
Cyphomyrmex sp4	10	3	1				6	4	3
Cyphomyrmex sp5	1					1	2		1
Cyphomyrmex sp6							2	1	
Cyphomyrmex transversus							1		
Mycocepurus smithii					1				
Ponerinae									
Dinoponera gigantea								3	
Hypoponera sp1									1
Odontomachus bauri	3								1
Pachycondyla harpax							1		
Pseudomyrmecinae									
Pseudomyrmex filiformis							1		
Pseudomyrmex gracilis								1	
Pseudomyrmex holmgreni							1		
Pseudomyrmex tenuis							1		
Pseudomyrmex termitarius		1		1	1		14	14	4

b) **Ants from the Forest**

Ant species	Forest-A	Forest-B	Forest-C
Dolichoderinae			
Dolichoderus sp3	1	0	0
Dolichoderus sp5	0	1	1
Ecitoninae			
Eciton sp1	1	0	0
Eciton sp2	0	0	1
Labidus sp1	1	1	4
Labidus sp2	0	0	2
Ectatomminae			
Ectatomma edentatum	1	3	4
Ectatomma lugens	16	18	8
Ectatomma tuberculatum	0	1	1
Gnamptogenys sp2	4	0	0
Gnamptogenys sp3	1	9	1
Gnamptogenys sp4	7	2	2
Gnamptogenys sp5	0	1	3
Formicinae			
Brachymyrmex sp2	1	2	1
Brachymyrmex sp5	2	0	2
Brachymyrmex sp6	0	0	1
Brachymyrmex sp7	0	0	1
Camponotus sp1	10	7	8
Camponotus sp4	1	1	1
Camponotus sp5	2	0	1
Camponotus sp6	2	0	0
Camponotus sp7	3	0	1
Camponotus sp8	0	1	0
Camponotus sp9	0	1	0
Camponotus sp10	0	0	1
Camponotus sp11	0	0	3
Camponotus sp17	1	4	0
Gigantiops destructor	0	2	1
Paratrechina sp1	2	2	0
Paratrechina sp2	0	0	1
Paratrechina sp4	1	1	0
Paratrechina sp5	1	0	1
Paratrechina sp6	1	0	1
Paratrechina sp7	0	0	3
Myrmicinae			
Acanthognathus sp1	0	1	0
Acromyrmex subterraneus brunneus	0	2	0
Atta cephalotes	0	0	9
Cephalotes atratus	0	0	1
Cephalotes oculatus	1	0	2
Cephalotes sp4	0	1	0

Cyphomyrmex sp1	0	1	0
Crematogaster sp1	5	1	0
Crematogaster sp2	0	1	1
Crematogaster sp3	5	7	11
Crematogaster sp4	0	2	1
Crematogaster sp5	0	0	2
Hylomyrma sp1	1	0	1
Megalomyrmex drifti	0	0	1
Myrmicocrypta sp1	1	0	0
Octostruma sp1	0	2	0
Pheidole sp1	2	1	4
Pheidole sp2	0	11	5
Pheidole sp3	4	1	2
Pheidole (gr. *tachigaliae*) sp6	3	0	2
Pheidole sp10	0	0	3
Pheidole sp11	2	3	5
Pheidole sp12	0	1	2
Pheidole sp14	0	2	0
Pheidole sp15	0	0	1
Pheidole sp16	0	1	0
Pheidole fracticeps	0	0	1
Pheidole sp21	3	5	2
Pheidole sp27	1	1	1
Pheidole sp28	1	1	1
Pheidole sp30	0	1	0
Pheidole sp33	1	0	0
Pheidole sp34	0	1	0
Pheidole sp36	0	0	1
Pheidole sp39	2	0	1
Pheidole sp40	1	7	0
Pheidole sp41	0	1	0
Pheidole sp42	0	1	0
Pheidole sp46	1	0	0
Pheidole sp47	1	2	1
Pheidole sp48	1	1	0
Pheidole sp49	0	1	2
Pheidole sp50	0	0	1
Pheidole sp53	2	3	1
Pheidole sp55	0	1	0
Pheidole sp56	1	0	1
Pheidole sp58	0	1	0
Rogeria sp1	1	0	0
Sericomyrmex sp1	6	6	4
Sericomyrmex sp2	0	1	1
Solenopsis sp1	1	6	2
Solenopsis sp2	4	2	1
Solenopsis sp3	1	5	0
Solenopsis sp6	0	0	1
Solenopsis sp7	0	0	1

Strumigenys sp1	0	0	1
Strumigenys sp2	4	1	3
Strumigenys sp3	0	0	1
Strumigenys sp4	0	0	1
Trachymyrmex (gr.*cornetzi*) sp1	1	0	0
Trachymyrmex sp2	0	3	0
Trachymyrmex relictus	0	1	0
Trachymyrmex ruthae	1	1	0
Trachymyrmex farinosus	0	1	2
Wasmannia auropunctata	0	1	0
Ponerinae			
Hypoponera sp1	1	0	0
Leptogenys sp2	0	1	0
Odontomachus (px. *haematodus*)	2	1	1
Odontomachus laticeps	2	0	0
Odontomachus sp3	5	2	1
Odontomachus sp4	1	0	0
Pachycondyla arhuaca	5	3	1
Pachycondyla sp2	5	8	6
Pachycondyla sp3	2	0	0
Pachycondyla sp4	5	3	4
Pachycondyla sp5	1	0	0
Pachycondyla sp6	0	0	2
Pachycondyla sp7	0	0	2
Pachycondyla striata	2	3	5
Pseudomyrmicinae			
Pseudomyrmex sp1	0	0	2
Pseudomyrmex sp3	0	0	2

CHAPTER II

SOYBEAN EXPANSION AND THE LOSS OF BIODIVERSITY IN THE AMAZON

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ABSTRACT

With the moratorium on conversion of Amazon forest to soybean plantations, cattle pastures in the Amazon region are increasingly being converted to soybean. In turn, forest is then cleared for new cattle pastures. Although the moratorium has stopped direct conversion the overall effect is minimal. Both soybean plantations and cattle pastures have huge effects on the regions vegetative and soil properties, which in turn will affect the great biodiversity of the region. Here we see how differences in these vegetative and soil properties between soybean plantations, cattle pasture, and a third alternative agroecosystem, agroforestery, affect ant biodiversity. We find that ant biodiversity is much lower in pastures and soybean agroecosystems, supporting almost half as many species as in the agroforestry system. Across all three agroecosystems sites with higher leaf-litter biomass and soil nutrient (K, Na and Ca) levels supported higher levels of ant biodiversity. Soybean plantations close to forest fragments had higher levels of ant biodiversity, but this effect was not seen in the other agroecosystems. Based on this work we suggest that if we are to preserve biodiverstiy in the face of agricultural expansion in the Amazon region we must attempt alternative agricultural systems, like agroforestry, instead of the currently used systems of soybean and cattle pasture.

Keyword: ant communities, land use, Amazon, environmental impacts, leaf litter, biodiversity, agroecosystem.

INTRODUCTION

Brazil is the world's second largest producer of soybeans, with 57.1 million tons harvested in 2009(1, 2). In addition to traditional demand streams, recent expansion of biodiesel production has also contributed to the expansion(3, 4). Together with cattle ranching, the proliferation of extensive soybean fields have pushed the agricultural frontier further into the Amazon forest (5).

In 2006 the Brazilian Association of Vegetable Oil Producers (6) and the Brazilian National Grain Exporters' Association (ANEC), major Brazilian soybean traders, announced a soybean moratorium, in which they agreed to a two-year moratorium on buying soybeans from deforested areas in the Amazon. The moratorium came after strong pressure from environmental groups and NGO pressure on retail and fast-food chains in Europe, as the links between European food products and the destruction of South American forests due to soybean expansion were becoming common knowledge.

In 2008 the moratorium was renewed for two more years with the addition of the Brazilian Environmental Ministry coming on board. The soybean moratorium is estimated to have contributed to an almost 50% reduction in deforestation in the Amazon. The total deforestation in Amazon was 20,644.29 during the period of 2000-2007, and according of the latest PRODES/INPE 2010 report covering the period of 2007-2009 the deforestation rate dropped to 11,578 km^2/year(7).

Although the moratorium has seen some success, more recently there seems to have been a change in the general dynamics of land use change. Partly because of the moratorium, it has become the norm to expand soybean production into areas formerly devoted to cattle pastures. This has led, in turn, to clearing new areas of Amazon forest to produce new cattle pastures, which are not covered by the moratorium. As a consequence, soybean production remains a major driver of deforestation in the Amazon region, although in an indirect fashion (3).

In addition to its direct effects on deforestation, the conventional soybean cropping system profoundly alters ecosystems (8). Soybean plantations in the Amazon are highly mechanized monocultures that require vast amounts of agrochemical and fossil fuel inputs, degrading the physical and chemical soil characteristics (9). To establish the plantations it is necessary to remove all native vegetation, disrupting most biological and biogeochemical cycles (10). After the removal of the native vegetation nearby forest areas become more susceptible to fire providing a positive feedback mechanism that increases deforestation (11). As only large-scale plantations are profitable, extensive areas are essential for the system and therefore the environmental impacts, including loss of biodiversity, are large. (8, 12).

Agroforestry systems (we will use the Portuguese abbreviation SAFS), have been suggested as an alternative and appropriate form of agriculture that can reduce the environmental impacts of agriculture in the Amazon (13, 14). SAFS do not represent a specific management type, but a general concept with many modalities in which annual crops are planted with perennial timber or fruit trees. Inputs, such as fertilizers, are used primarily during the first year or two on annual crops, like corn or rice, and as the trees are initially established. Subsequently, the amount of inputs needed is relatively low. Currently, the amount of land in SAFS in the Amazon is very small compared to that used for soybean or pasture (15).

The Amazon region is of global importance to biodiversity conservation, containing a quarter of all described terrestrial species in the world (16). Increasingly, this hotspot of biodiversity is being converted to cropland and pastures, causing the loss of biodiversity. With the increasing role of soybean in the Brazilian economy, it is expected that soybean production areas will be maintained or even expanded. Whether this expansion takes place at the expense of forests directly or indirectly by replacing cattle pasture areas and displacing them into the forests, it is imperative that we understand how these two agricultural systems affect the region's biodiversity. Likewise, it is important to evaluate alternative systems that may provide a livelihood for the people of the Amazon and, at the same time, maintain biodiversity. Recent predictive models have been useful in analyzing biodiversity losses of animals and plants by cropland expansion (17-20). Here we expand on that work and compare the diversity found in soybean plantations, pastures and SAFS. The first two systems represent the majority and most rapidly growing types of land use in the Amazon region, and the third represents an alternative agricultural system for the region.

We use the ant community as a model group to assess the impacts of these different land uses. We also examine the main environmental variables that can support more ant biodiversity within each agricultural system, and using predictive bayesian models we predict species richness based on various environmental variables.

METHODS

We collected ants using pitfall traps in soybean plantations, pastures and SAFS in Pará state of Brazil during the rain season, January to April in 2008 (for more details see Supplementary Information at the first paper). In each site we measured a number of environmental, vegetation and soil variables. We compared ant species richness using the expected species accumulation curves (Mao Tao) adjustment in EstimateS (21), and then employed a Predictive Bayesian Model to evaluate which of our environmental, vegetation and soil variables best explained ant biodiversity across the three agroecosystems (see environmental variables predication model, EVPM in Supplementary Information at the first paper for full explanation).

RESULTS

Soybean plantations and pasture sites had lower species richness than that found in SAFS sites (Fig. 1). Overall, pastures had 44% and soybeans 29% of the species richness of SAFS. In addition there were 41 ant species found exclusively in SAFS sites, compared to just four species unique to pasture and five to soybean, the latter being well-known from human-influenced environments (see Supplementary Information Table 2 (a) at the first paper for ant species list from agroecosystems).

Predictive Bayesian Model: Environmental Variables

Many environmental variables were strongly correlated with ant species richness when examining all systems together (Fig. 2). Plant species richness, plant density and pH were negatively correlated with ant species richness, while leaf-litter biomass, organic matter, and several soil nutrients (i.e., Nitrogen, Potassium, Sodium, and Calcium) had a positive effect on ant biodiversity. We estimated ant species richness as a function of the sample size and the environmental variables (see Supplementary Information at the first paper, Table S1). We used the parameters from the environmental variables prediction model (EVPM) (Fig. 3) to investigate how ant species biodiversity would be affected under different conditions for two selected environmental variables: *herbaceous plant density and leaf-littler biomass* (see Supplementary Information at the first paper for full explanation) (Fig. 3). Predictions show that pastures closer to forest fragments had more ant biodiversity than soybean farms, but when both were more distant from fragments this difference disappears (Fig. 3a). On the other hand SAFS had higher ant species richness than soybean or pastures no matter how far from a forest fragment. For low or high plant density SAFS had considerably higher ant species richness than either soybean or pasture, which had roughly the same ant species richness in both low and high plant density (Fig. 3b). Leaf-litter biomass had the largest effect on ant biodiversity across all systems, doubling the number of ant species observed. Still in both low and high leaf-litter biomass conditions SAFS has the highest ant biodiversity (Fig. 3c).

DISCUSSION

We observed dramatic differences in ant species richness between the three agroecosystems; SAFS supports more than twice the number of ant species as the other two systems, soybean and pasture. In addition there were 41 ant species found only in SAFS systems and the few species found only in soybean or pasture and not SAFS are those commonly associated with highly human-influenced environments. Our study demonstrates that agroecological systems that include trees, SAFS, drastically contrasted with crop and pasture areas in terms of biodiversity. Further we observed that replacing even a small fraction of either pasture or soybean with SAFS will result in a significant increase in ant species richness.

The EVPM showed that a number of other environmental variables were important in explaining ant biodiversity, but for any set of environmental conditions SAFS still had higher ant biodiversity than the other systems. Furthermore, our EVPM showed that pasture systems close to forest fragments had higher ant species richness than pasture systems farther away from forest fragments. This highlights the importance of maintaining forest fragments near farms as a source of ant species in these agroecosystems.

Despite the soybean moratorium, land is still being converted to soybean in the Amazon. Generally, the conversion goes from forest to pasture and from pasture to soybean (6). Here we have shown that soybean plantations as well as cattle pastures, the two most common and rapidly expanding land use systems in the Amazon, support little associated biodiversity. Agroforestry systems will not only support higher levels of resident biodiversity, but will also provide a habitat that is more likely to support migration between forest fragments (22).

Currently cattle ranching and large-scale monocultures, such as soybean, are rapidly displacing forest in the Amazon region (6), according to our results if this land were instead used for SAFS agroecosystems the loss in biodiversity would be much less.

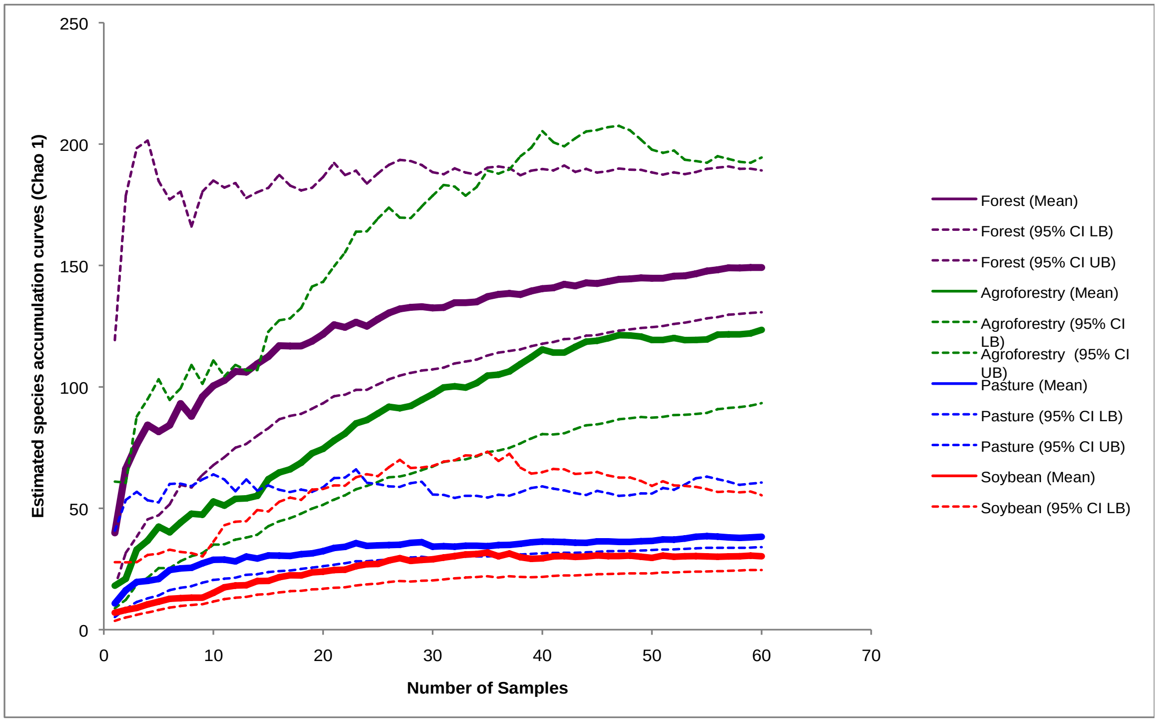


Figure 1: Ant species richness (95% CI) for Agroforestry (SAFS), pasture and soybean agroecosystems.

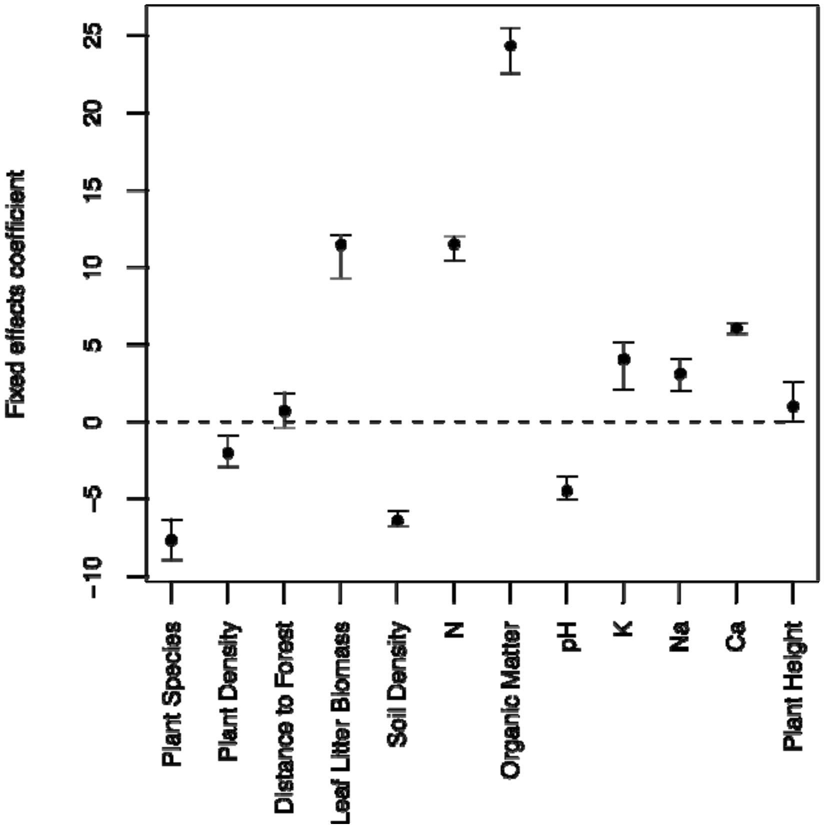
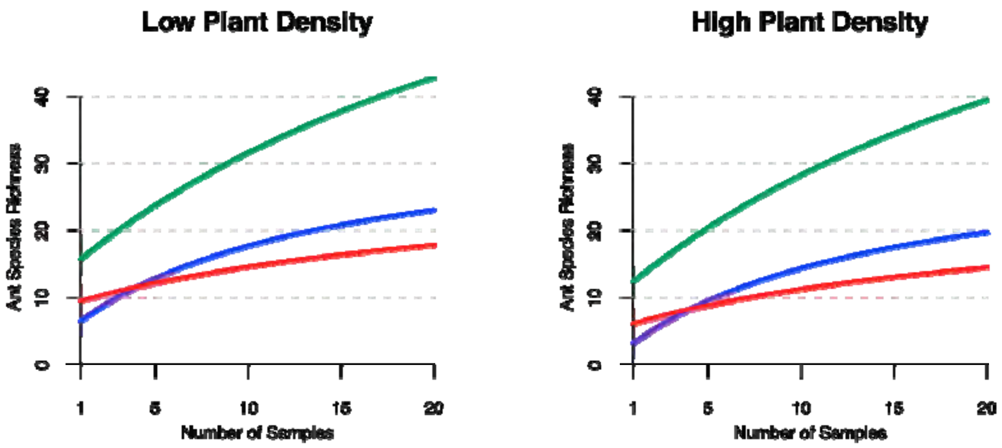


Figure 2: Effect of environmental variables on ant species richness. Here we do not considered systems, just the total effect of each environmental variable.

A.



B.

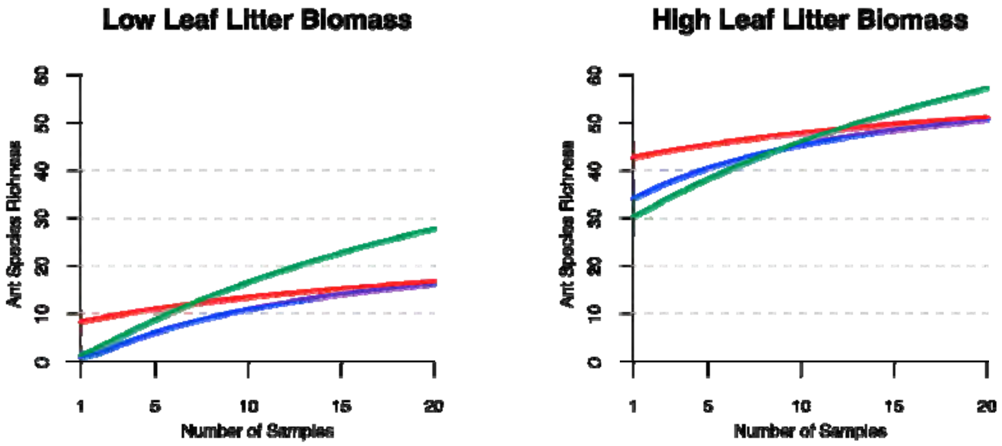


Figure 3: Predicted species accumulation curves by our EVPM. Using this model we predict species accumulation under high and low conditions for two environmental variables at each of the three agroecosytems. In each graph green is agroforestry (SAFS), blue is pasture and red is soybean. The shaded region shows the 95% predictive interval of the number of species. A) Shows the results for low and high plant density. And B) shows the results for low and high plant biomass (Agroforestry = green line; Pasture = blue line; Soybean = red line).

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CHAPTER III

Running head: Ant biodiversity loss in forest plantations

THE IMPACT OF FOREST PLANTATIONS OF NATIVE AND EXOTIC SPECIES ON ANT BIODIVERSITY IN THE AMAZON REGION

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ABSTRACT

In recent years the land devoted to forest plantations has increased in Brazil due to the twin incentives of joining the international timber market and acquiring carbon offsets. This change in land use is of particular importance because of the region's high level of biodiversity and importance in climate regulation. Therefore, it is important to understand the effect these plantations will have on local biodiversity, especially in biodiversity hotspots such as the Amazon region. Here we compare ant biodiversity in timber plantations planted with paricá (Schizolobium amazonicum), a native tree, and teak (Tectona grandis) an exotic tree, both planted as timber trees in the Amazon region. We also examined environmental variables that explain ant biodiversity across the two plantation types. We found almost double the number of ant species in the native plantations as compared to the exotic plantations (60 species compared to 35), and 29 species unique to the native plantations compared to just four in the exotic plantations. Chao1 estimates of species richness and species accumulation curves using Mao Tao estimates from observed richness confirmed this difference. Leaf litter biomass, which is typically correlated with ant biodiversity, had a positive effect on ant biodiversity in the native plantations, but no effect in the exotic plantations.

Key words: *Tectona grandis, Schizolobium amazonicum,* Amazon Region, Forest Plantation, Ant biodiversity

INTRODUCTION

Global warming has been identified as one of the most pressing and challenging problems that we face in the Twenty First Century (Asner et al. 2005; Feeley & Silman 2009; Hubbell et al. 2008; Malhi et al. 2008). Since deforestation has a strong impact on climate change, not only locally, but also at regional and global scales (Fearnside et al. 2009; Foley et al. 2007; Malhi et al. 2008; Sampaio et al. 2007), among the strategies that have been proposed to mitigate global warming is the reduction of deforestation, with strong pressure on countries that have extensive tropical forest such as Brazil, and some countries in Africa and Asia (Broadbent et al. 2008; Feeley & Silman 2009; Michalski et al. 2008; Nepstad et al. 2006). A complementary strategy has been the creation of a carbon market through which countries receive incentives (monetary compensation) for their carbon uptake (ICBE 2010). In combination with an increasing demand for timber worldwide this has led to an increase in tree plantations (FAO 2009).

Although such plantations can play an important role in carbon uptake (Kaul et al. 2009; Kirby & Potvin 2007; Kraenzel et al. 2003) they may also have a negative impact on biodiversity (Attignon et al. 2005; Attignon et al. 2004; Gardner et al. 2008; Watt et al. 2002). With the increasing importance of plantations for global warming mitigation, it is important to consider potential negative impacts that may offset the expected benefits. For example, the negative effect of timber monocultures on the diversity of several groups of animals has been well-documented (Attignon et al. 2005; Gardner et al. 2008; Lachat et al. 2006) and plants (Healey & Gara 2003). Also, these effects may be worse when exotic tree species are used (Healey & Gara 2003; Lachat et al. 2006).

The Amazon is the largest reservoir of biodiversity in the world, home to more than 40,000 described vascular plant species (Feeley & Silman 2009), more than half of the described animal species (Capobianco et al. 2001; Dirzo & Raven 2003), and containing major river basins with an extraordinarily high diversity of fresh water organisms. It also influences global climatic stability (Avissar & Werth 2005; Gibbs et al. 2007). As such, the Amazon region plays a major role in the two main environmental problems we are facing currently -- biodiversity loss and climate change. Brazil, though still a minor player in the international timber trade (FAO 2009), has seen an increase in timber plantations in several states in the last few years (ABRAF 2009), increasingly located in the Amazon region. Brazil's timber plantations are almost exclusively planted with exotic species, mainly *Eucalyptus* and *Pinus*, and rather recently, *Tectona grandis* (ABRAF 2009), although *Schizolobium amazonicum*, a native tree, is also planted in the Amazon region. *Tectona grandis,* teak, is one of the most profitable woods on the international market, and as such is increasingly used in timber plantations in Brazil and is quickly replacing native-tree plantations in the Amazon region.

It is important to note that although teak has a wide distribution in tropical regions, in the Amazon it is an exotic species and is almost exclusively planted in monocultures. A few studies have documented the impact of teak-planted forests on biodiversity in the Americas, Africa and Asia (Attignon et al. 2005; Healey & Gara 2003; Jenkins et al. 2003; Lachat et al. 2006), but little is know about it's impacts within biodiversity hotspots of the Americas, such as the Amazon region. Likewise, little is know about the environmental variables that may impact biodiversity in teak plantations. In spite of the recent increase in the establishment of large monocultural plantations of exotic species in the Amazon region

(ABRAF 2009), no study has been conducted to assess their impact on biodiversity or to compare exotic and native forest plantations. In this work we aimed to evaluate the impact caused by exotic-tree plantations compared to native-tree plantations on ant biodiversity in the Amazon.

Ants are frequently used as indicators of biodiversity in the tropics because they are sensitive to anthropogenic habitat disturbances, they have ecological interactions with many other biotic components of terrestrial ecosystems and they are relatively easy to sample and identify (Andersen et al. 2004; Andersen et al. 2002; Armbrecht et al. 2005; Floren & Linsenmair 2005; Roth et al. 1994). We used teak (*T. grandis*) as our exotic species because of its increasing role in timber plantations throughout the tropics (FAO 2009), and especially in the Amazon region (ABRAF 2009), and paricá (*S. amazonicum*) as our native species because it is one of the most common native trees planted for timber in the region. This study also examines which environmental variables best explain ant biodiversity in native- and exotic-planted forests.

METHODS

We collected ant samples in three monocultural plantations of paricá, the native species, and three monocultural plantations of teak, the exotic species. The paricá plantations were seven years old and the teak plantations ten years old, both had the same density of trees, one tree per 4m x 3m area. Our collection took place between January and March 2008 in the districts of Capitão Poço, Garrafão do Norte and Irituia, Pará in Brazil. Paricá (*S. amazonicum*) is a non-deciduous leguminous tree with small leafelets. Teak (*T. grandis*), on the other hand, has large, deciduous leaves and makes up all of the forest tree plantations in the northeastern region of the State of Pará, Brazil.

Ant Sampling

We collected ants with pitfall traps (diameter of 18 cm and a height of 10.5 cm). In each area of collection, 20 traps were installed 15 m apart and arranged in four rows of five pitfalls. Each was left open for 48 hours. We transferred the material we collected to jars filled with 70% ethanol, and transported the samples to the Pheromones and Insect Behavior Laboratory of the Federal University of Viçosa, in Viçosa, State of Minas Gerais, Brazil. The material was then sorted, mounted, and identified to species or morphospecies based on the literature (Bolton 2006; Hölldobler et al. 1990) and in comparison with specimens of the Formicidae Collection of the Museum of Zoology of the University of São Paulo.

Environmental Variables

We collected a set of environmental variables to examine their role in determining ant species diversity inside each forest plantation. We divided the variables into two sets, vegetation and soil chemical variables. These two sets of variables were then used to build two statistical models to examine their effects on ant species diversity.

*Vegetation variables***:** We collected the vegetation variables in a 1-meter radius from the center of the collecting pitfall trap. We measured the following variables in the herbaceous layer (up to 1.50m): plant species richness (sp), plant density (dp), and average plant height (altp). In addition we measured distance from the nearest forest (dis), quantity of litter biomass (lb) and soil depth (ds). We determined the richness and density of the herbaceous plant layer by collecting, counting, and identifying the morphospecies of all plant species within a meter radius of each pitfall trap. We estimated the average plant height by measuring the heights of up to 10 plants within the 1-meter radius. Leaf litter biomass was estimated by throwing a 0.2 m X 0.2 m square frame within a 2 m radius from the center of the pitfall trap and collecting all litter material within the square. The biomass was quantified in grams (g) by weighing the dry material. Finally, we recorded the distance (m) from each pitfall trap to the nearest native forest fragment.

Soil chemical variables: We measured the nitrogen (N%), organic matter (MO [g/kg]), pH (water), phosphorous (P[mg/dm^3]), potassium (K[mg/dm^3]), Calcium (Ca [cmol$_c$/dm^3]), the base Calcium-Magnesium (Ca+Mg [cmol$_c$/dm^3]), and Aluminum (Al [cmol$_c$/dm^3]). For all of the soil chemical variables we collected and pooled ten 100 cm^3 soil samples around each trap (each of the ten samples from within 2 meter of the trap). All soil collecting process and chemical analyses were conducted according to the Brazilian Agricultural Research Corporation (EMBRAPA) soil protocol (EMBRAPA-Solos 1997).

Statistical Analyses

Biodiversity of Ants

We compared ant species richness between native and exotic-planted forests using smoothed species accumulation curves with 95% confidence intervals and species richness estimates. We computed the sample-based accumulation curves for ants with the Mao Tao function using EstimateS 7.5 (Colwell et al. 2004). Estimates of ant species richness for native and exotic-planted forests were also obtained from EstimateS 7.5 using the first order Chao1 (Colwell et al. 2004). This estimator is based on estimated variance, and the corresponding log-linear 95% confidence intervals and re-sorting the species frequency instead of the abundance; it is a nonparametric estimator that performs relatively well under a wide range of sample sizes (Colwell & Coddington 1994).

The effects of environmental variables on ant biodiversity

We used the environmental variables, both herbaceous-layer vegetation and soil chemical composition, as a measure of environment structure to see how those variables affect ant diversity in both native- and exotic-planted forests.

Then, we built a full model in which the observed ant species richness served as the predicted variable and the environmental variables as the explanatory variables. We separated the vegetation and soil chemical variables into different models individually and included the double interactions in each:

Where:

- AS is the ant species richness
- s is a binomial variable, teak (t) or paricá (p)
- EV are the vegetation or soil chemical variables, and the a_i and $b_{i,j}$ coefficients are the parameters of the model.

The variable *forest plantation* (s) is included in the model so as to differentiate between the types of native- and exotic-planted forests. We also analyzed the effect per each kind of forest, and in these cases, we removed the variable *forest plantation* from the statistical model and analyzed each different forest plantation to make sure that the effect was caused by the explanatory variables and not just an effect caused by forest plantation type. We selected the best model using *stepwise regression* (stepAIC) because that method allows the introduction and removal of variables in different positions of the model until it reaches the most parsimonious minimum adequate model (Venables & Ripley 2002). We used the Akaike’s Information Criterion (AIC) as proposed by (Whittingham et al. 2006). However, we altered the parameter k of the function stepAIC from 2.71 (p = 0.1) to 3.84 (p = 0.05), in order to increase the strictness in the criterion to add and remove variables in the model. We utilized Generalized Linear Models as the estimating method with distribution of errors Poisson because we had count data (Crawley 2009).

We submitted all of the variables of the last model to a Deviance analysis, and only those variables that in the model were significant at a 5% level or enrolled in significant interactions were maintained. After the selection of the last model, we used the residues analysis to verify the adequacy of the model and the distribution of the errors used. All the analyses were done using R (Bolker 2008; Crawley & Wiley InterScience (Online service) 2007) and Estimate S (Colwell & Coddington 1994; Hornik 2009).

RESULTS

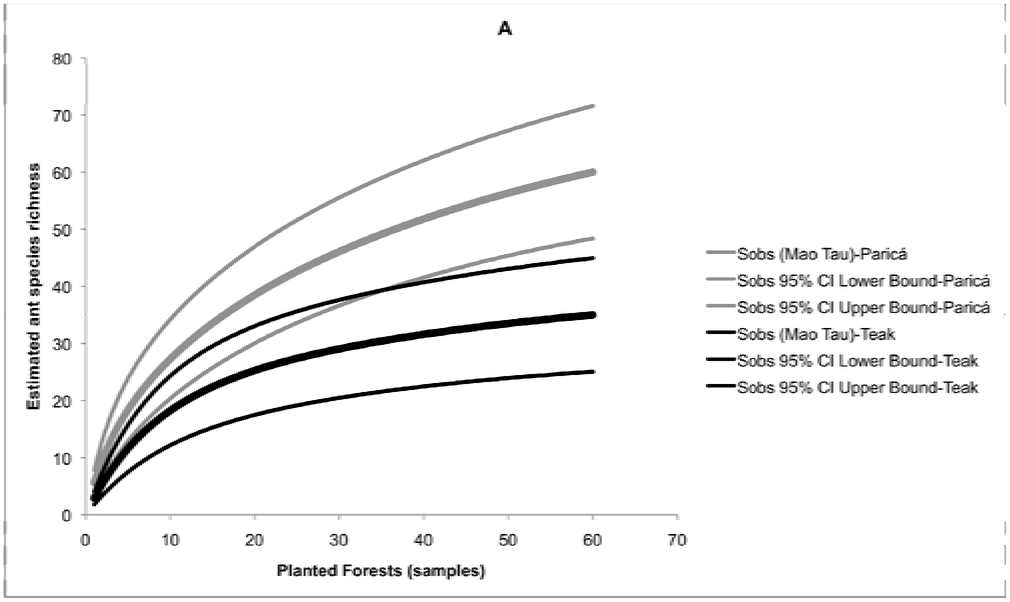
We collected an average of 6.3 and 3.01 ant species per trap in paricá and teak, respectively, belonging to a total of 64 species, 17 genera and six subfamilies. Of these 64 species 29 were collected exclusively in paricá sites and only four exclusively in teak sites (see supplementary material S1 for a complete species list by site). The native plantations (paricá) had almost double the ant species richness as the exotic (teak) plantations (60 and 35 ant species, respectively). From 120 traps in all plantations just one trap was empty in a paricá plantation, while five traps were empty in teak.

The Mao Tao observed species with 95% confidence intervals shows this to be a statistically significant difference (Fig. 1-A). Likewise the Chao1 estimate indicates an almost doubling of the ant species richness in the paricá plantations as compared to teak (Fig. 1-B). Even when considered separately, each paricá site had more ant species than each teak site (Fig. 2).

The type of forest plantation affected ant species richness significantly ($p < 0.005$, Table 1). In addition vegetation and soil variables also impacted ant species richness (Table 1). Although teak forest plantations have a closed canopy, deciduous leaves and four-year head start over the paricá plantations, they had less leaf litter biomass than the paricá plantations ($mean_{par}$=133.405g/sample; $mean_{teak}$=122.145g/sample).

Furthermore, leaf litter biomass had a very important effect on determining ant species richness when all sites where considered together ($p_{lb} < 0.05$) (Table 1, Fig. 3). But when we considered the systems separately, leaf litter biomass had a positive effect on ant species richness only in Paricá plantations, and no significant effect in Teak plantations (Table 1). Similarly, the teak plantation that was nearest to a native forest fragment associated with high plant species richness had higher ant species richness ($p_{dist*sp} < 0.05$). Some variables considered jointly also had an effect. Specifically, the interaction of plant richness and soil depth ($p_{sp*ds} < 0.05$) (Fig. 4-a) and the interaction of plant density and leaf litter biomass ($p_{dp*lb} < 0.05$) (Fig. 4-b) showed a strong influence on ant species richness.

Soil nutrition, in particular the amount of calcium ($p_{Ca} < 0.0005$), also played an important role in ant species richness when the two forest plantations were considered together (Table 1). Other nutrients and interactions showed marginal levels of statistical significance: potassium marginally negatively affecting ant species richness ($p_K = 0.067$) and the interaction of nitrogen and pH affecting it positively, while the interaction of nitrogen and calcium-manganese negatively affecting species richness ($p_{N*pH} < 0.05$, and $p_{N*CaMg} < 0.05$) (Table 1).



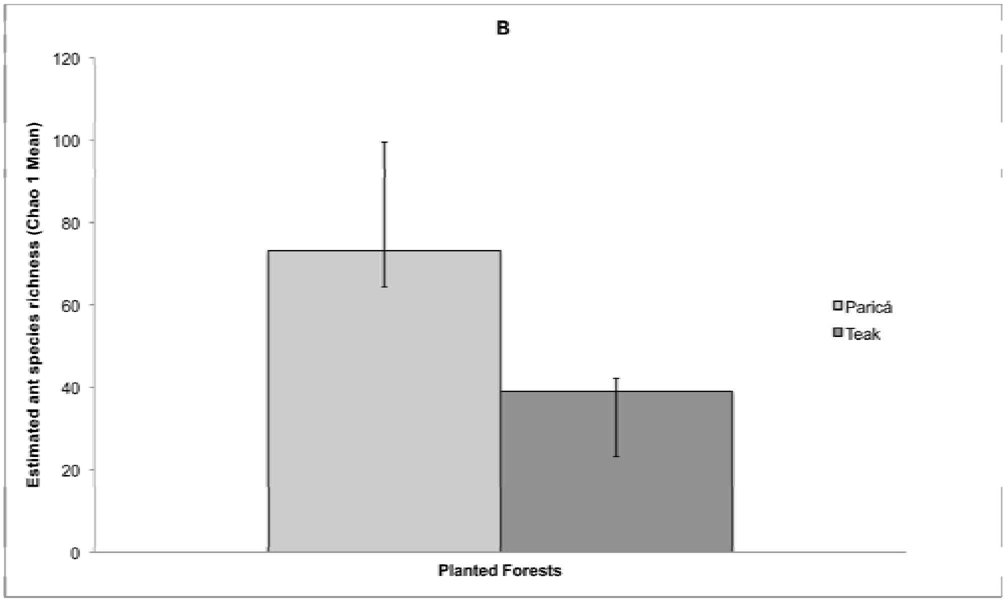


Figure 1: Ant species richness (95% CI) for native- and exotic-planted forests where A is the observed species smoothed accumulation curve (Mao Tao 95% CI) and B is the estimated ant species richness (Chao 1 95 CI).

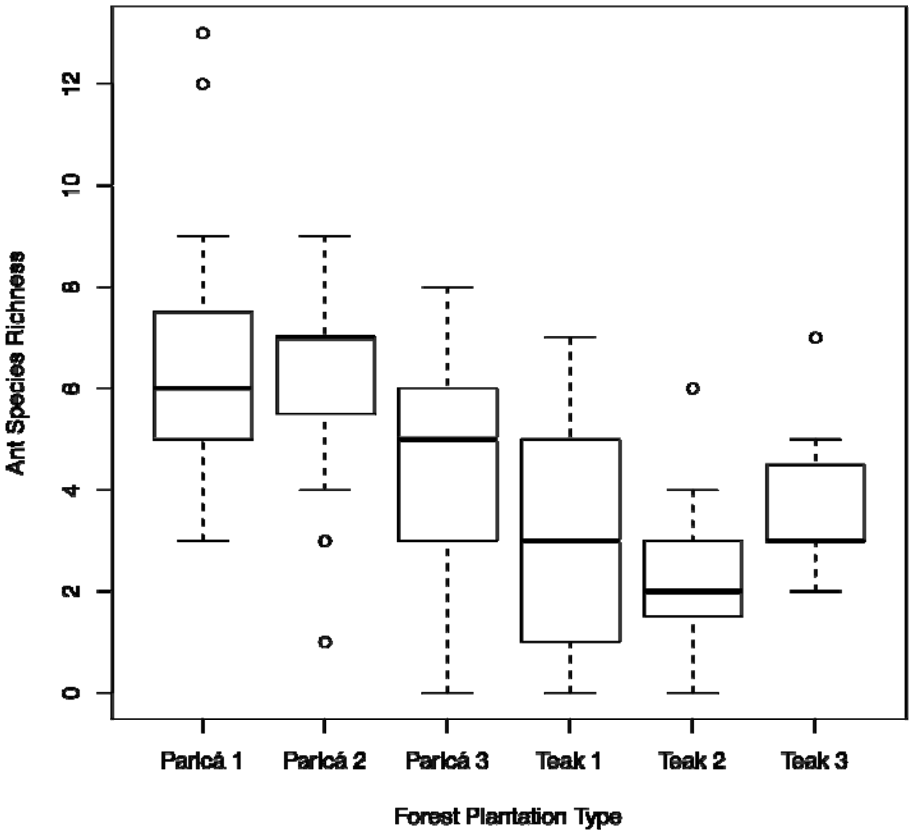


Figure 2: Ant species richness in Paricá- and Teak-planted forests.

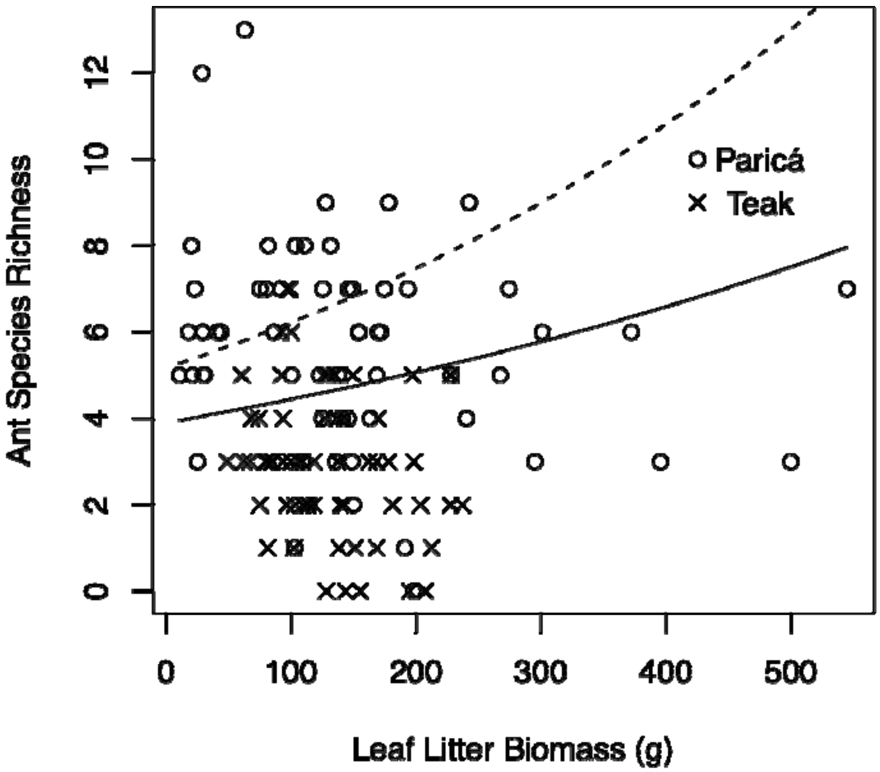
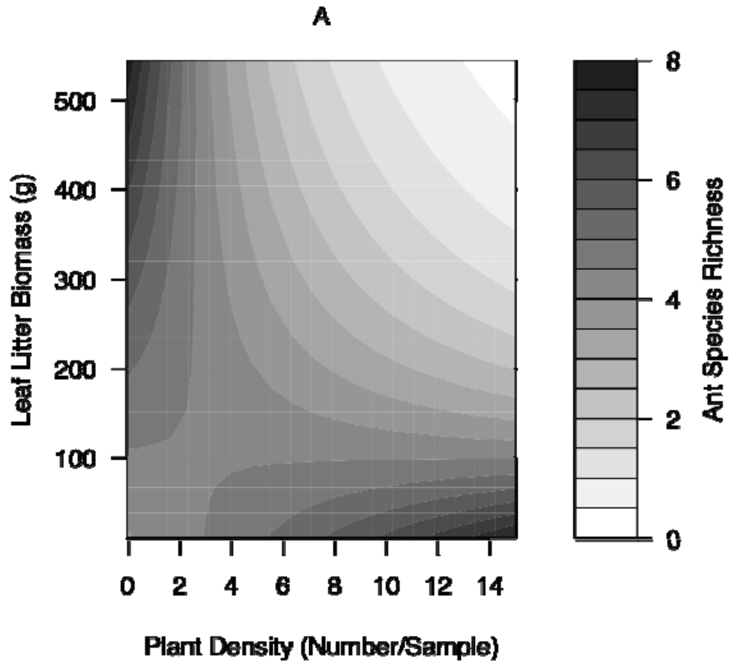


Figure 3: Ant species richness by leaf litter biomass. The relationship is significant for both forest plantations put together, indicated by the solid line, and when considering just the Paricá plantations on their own, the dotted line.


A
Leaf Litter Biomass (g)
500
400
300
200
100
0
2
4
6
8
10
12
14
Plant Density (Number/Sample)
8
6
4
2
0
Ant Species Richness

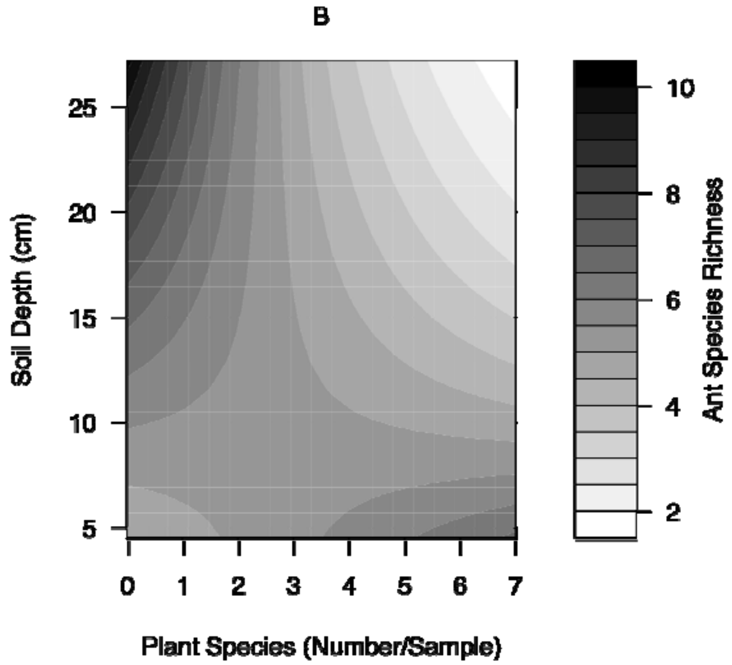

B
Soil Depth (cm)
25
20
15
10
5
0
1
2
3
4
5
6
7
Plant Species (Number/Sample)
10
8
6
4
2
Ant Species Richness

Figure 4: The two significant interactions terms in our model. Figure 4A shows that plant species and soil depth on their own support higher ant species richness, but together lower ant species richness. Figure 4B shows the same result for plant density and leaf littler biomass in paricá and teak planted-forest.

DISCUSSION

Ant species richness was significantly higher in plantations planted with the native species *S. amazonicum* (paricá), than in plantations planted with the exotic *T. grandis* (teak). This was true not only for the observed species richness but also for the Chao1 estimate and also for all study sites separately (Fig. 1 and 2). Furthermore, vegetation and soil variables as well as distance to the nearest forest fragment, played a very important role in supporting high ant species richness in both planted forest.

The variable that had the strongest influence on ant species richness was leaf litter biomass (Fig. 3). However, other variables, specifically plant species in combination with soil depth (Fig. 4-a), plant density in combination with leaf litter biomass (Fig. 4-b), and amount of calcium (Fig. 5), also strongly influenced ant species richness.

Canopy and understory vegetative structure are important in determining species richness and community composition (Blaum et al. 2009). Highly heterogeneous and complex environments can support more species (Armbrecht et al. 2004; Correa et al. 2006; Ribas et al. 2003; Silva et al. 2007) by offering a wider range of resources, shelter from predation and foods, and additional required conditions, such as humidity, luminosity and temperature for ant communities (Fellers 1989; Lassau & Hochuli 2004; Levings 1983; Torres 1984). Even though the exotic-planted forests in our study were older and had a more closed canopy than the native-planted forests, their understory had a lower environmental heterogeneity, with less herbaceous plant species diversity, density of plants and less leaf litter biomass than the native planted forest. We expected that because teak is deciduous, these plantations would have more leaf litter biomass. However, this was not the case. We hypothesize that this lower environmental heterogeneity and less specious herbaceous plant community are responsible for the lower ant biodiversity, just over half as many species, in the teak plantation compared to the paricá plantation. Finally we suspect that ant abundance is lower in teak plantations because we had various empty traps, compared to one in the paricá sites.

In the native-planted forest leaf litter biomass was the environmental variable that best predicted ant biodiversity, with higher biodiversity in sites with more leaf litter. This result has been seen in other studies of invertebrate biodiversity (Armbrecht & Perfecto 2003; Armbrecht et al. 2005). But in the exotic-planted forest there is no significant relationship between leaf litter biomass and species richness (Fig. 3). Healy and Gara (2003) suggest that teak leaf litter suppresses seed germination because of a phenolic foliar leachate. This potential effect of teak leaf litter on the herbaceous community may explain why we do not see an effect of leaf littler biomass on ant species richness in teak, as we do in other systems (Armbrecht & Perfecto 2003; Armbrecht et al. 2005).

Other studies have found lower biodiversity in teak plantations than in mixed-planting forests (Attignon et al. 2005; Attignon et al. 2004; Grimbacher et al. 2007). This has been shown in both teak's native range as well as in other tropical regions, for both invertebrate communities (Attignon et al. 2005; Attignon et al. 2004; Jenkins et al. 2003) and understory, herbaceous communities (Healey & Gara 2003). This is the first report of it in the Amazon and the first to compare teak and another monocultural plantation of a native species.

In the exotic-planted forest there was a significant relationship between ant species richness and the interaction between leaf litter biomass and distance from the nearest forest fragment. This suggests that for plantations of this exotic species the native forest fragment may play an especially important role in maintaining ant species richness by acting as a source area out of which ants can migrate.

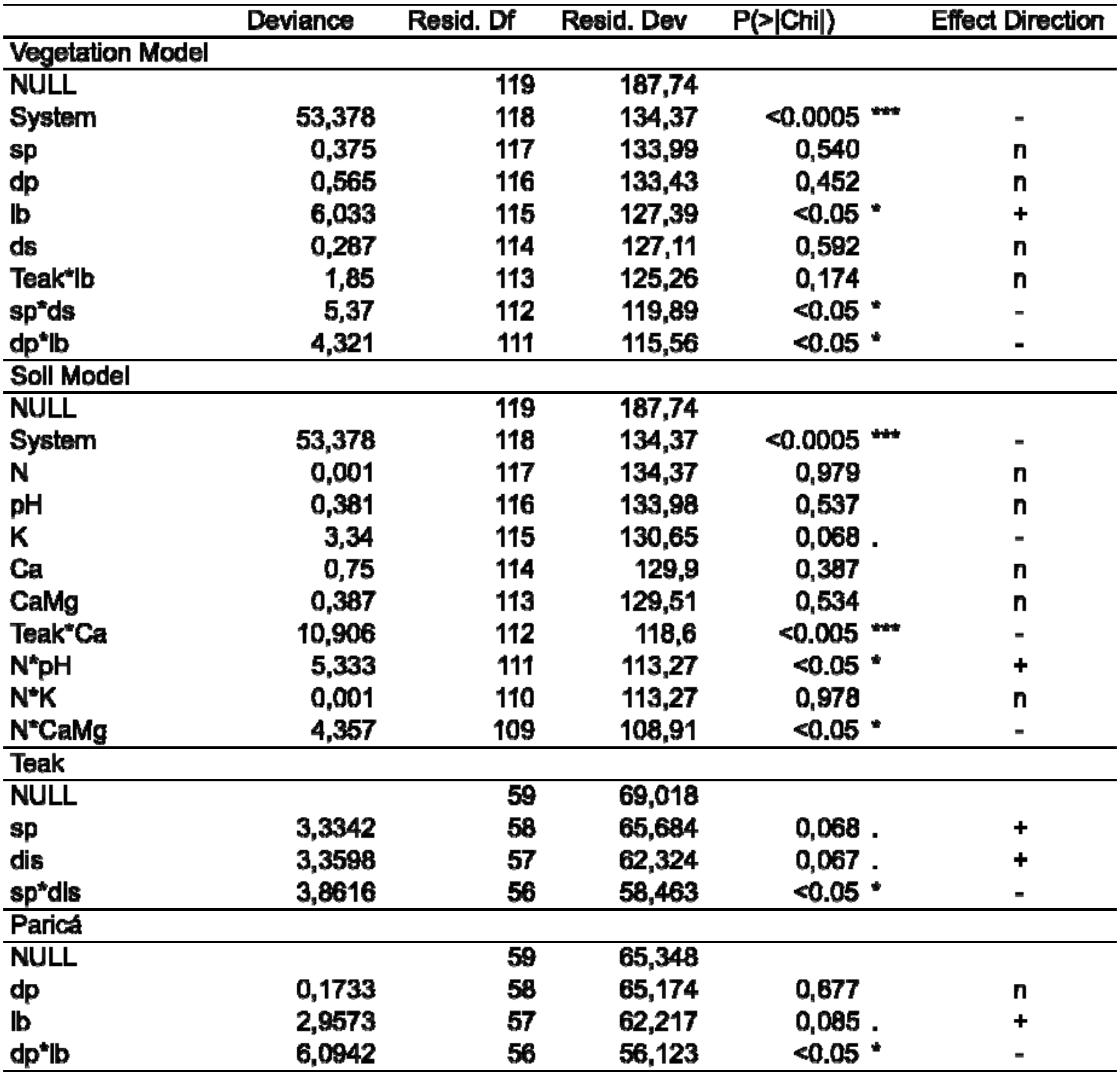
The question remains as to whether these plantations represent a high quality matrix (Perfecto & Vandermeer 2002; Perfecto et al. 2009; Vandermeer & Carvajal 2001). Which is to say are they able to maintain populations of ant species migrating from the forest fragments, or if they represent a sink or dead end for those species. If they can indeed maintain these ant populations, at least temporarily (until queens are produced and can fly away and establish a new colony in another forest fragment), it would be important to maintain numerous and large native forest fragments nearby exotic-planted forests and decrease the size of these exotic-planted forests to prevent regional species loss. However, given the low number of species found in the teak plantations, it seems likely that these plantations are of very low quality and may serve more as a sink to populations of forest species than a viable habitat, even if temporarily so.

The paricá plantations, due to the high number of species collected from them, seem to represent a higher quality matrix that can maintain populations of many forest species. However, since we did not collected data on ant richness within the forest fragments, we do not know how these plantations compare, in terms of species richness and composition, with the forest fragments. Nonetheless, native-planted forests' ability to maintain a higher level of ant biodiversity than exotic-planted forests may also impact nearby native forest fragments. If ant species can survive in these plantations they may be able to use them as a migration matrix between native forest fragments and thus offset inevitable local extinctions in native forest fragments (Perfecto & Vandermeer 2008). Thus planting forest plantations with native trees, and preferably a polyculture of native species, rather than exotic tree monocultures may impact not only the local biodiversity within these plantations, but also adjacent forest fragments and thus increase biodiversity on a regional scale (Perfecto & Vandermeer 2008; Vandermeer et al. 2010).

The Amazon region faces two great problems, climate change and biodiversity loss (Asner et al. 2005; Feeley & Silman 2009; Hubbell et al. 2008; Malhi et al. 2008). We must be cautious of solutions to one of these problems that exacerbate the other. Monocultures of teak plantations are such a solution, and they are gaining popularity as compensations for carbon credits become more widespread in the region.

In this study we document the low diversity of ant species in these plantations as compared to plantations of the native paricá. Teak monocultures sequester carbon and have become an economically attractive (ABRAF 2009) option in the Amazon region. However, they support much lower levels of biodiversity compared to paricá forests or mixed plantations (Grimbacher et al. 2007). The Amazon is one of the global hotspots for biodiversity so it is particularly important that, in attempt to mitigate climate change we don’t implement strategies that will reduce biodiversity. This study helps us to understand the potential tradeoffs between carbon sequestration and the loss of biodiversity in the Amazon region.

Table 1. Results from the step-wise model showing the effect of environmental variables on ant biodiversity, first on the complete data set and then for the two forest plantations separately. The abbreviations are as follows: sp = number of plant species, dp = plant density, lb = leaf-litter biomass, ds = soil depth, and dis = distance to closest forest fragment.



	Deviance	Resid. Df	Resid. Dev	P(>|Chi|)		Effect Direction
Vegetation Model						
NULL		119	187,74			
System	53,378	118	134,37	<0.0005	***	-
sp	0,375	117	133,99	0,540		n
dp	0,565	116	133,43	0,452		n
lb	6,033	115	127,39	<0.05	*	+
ds	0,287	114	127,11	0,592		n
Teak*lb	1,85	113	125,26	0,174		n
sp*ds	5,37	112	119,89	<0.05	*	-
dp*lb	4,321	111	115,56	<0.05	*	-
Soil Model						
NULL		119	187,74			
System	53,378	118	134,37	<0.0005	***	-
N	0,001	117	134,37	0,979		n
pH	0,381	116	133,98	0,537		n
K	3,34	115	130,65	0,068	.	-
Ca	0,75	114	129,9	0,387		n
CaMg	0,387	113	129,51	0,534		n
Teak*Ca	10,906	112	118,6	<0.005	***	-
N*pH	5,333	111	113,27	<0.05	*	+
N*K	0,001	110	113,27	0,978		n
N*CaMg	4,357	109	108,91	<0.05	*	-
Teak						
NULL		59	69,018			
sp	3,3342	58	65,684	0,068	.	+
dis	3,3598	57	62,324	0,067	.	+
sp*dis	3,8616	56	58,463	<0.05	*	-
Paricá						
NULL		59	65,348			
dp	0,1733	58	65,174	0,677		n
lb	2,9573	57	62,217	0,085	.	+
dp*lb	6,0942	56	56,123	<0.05	*	-

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CHAPTER IV

Running head: Ants diversity in orange orchards

EFFECTS OF ORGANIC AND CONVENTIONAL ORANGE ORCHARD MANAGEMENT ON ANT DIVERSITY IN THE AMAZON

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ABSTRACT

The biggest threat to biodiversity in the Amazon is the expansion of cattle ranching and agriculture. However, studies have shown that some agroecosystems can sustain elevated biodiversity. The goal of this study was to evaluate ant species richness and composition in orange orchards under organic and conventional management and to investigate what environmental variables influence ant diversity within these systems. Ants were collected in Pará State, Brazil. Twenty pitfall traps were installed in three replicates per management type. Vegetation and other environmental variables were measured and differences in ant species richness were examined using smoothed species accumulation curves. Overall, eighty-five ant species were collected; 69 in organic and 65 in conventional orchards. Smoothed species/sample curves show no significant differences in species richness between systems, nor in terms of percent of shared species within the two types of orchards. However, species overlap between orchards was less than 50%, and there were significantly more “pest” species in the conventional orchards. Under conventional management, plant species richness and plant density had a significant effect on ant species richness, while in organic orchards leaf litter was the most important explanatory environmental variable for species richness. Contrary to expectations, ant species richness increased with distance to the forest after other variables were taken into account.

Keyword: ant communities, orange orchard, distance from forest, environmental impacts, leaf litter, biodiversity, agroecosystem.

INTRODUCTION

Early in the 1970's the Amazon basin was the site of aggressive development projects, resulting in the region becoming the new frontier for Brazilian agriculture (Carvalho, Barros, Moutinho & Nepstad 2001; Martins, Praxedes, Santos, Silva & Costa 2007). Many of these projects brought with them uncontrolled deforestation and other negative environmental impacts (Carvalho et al. 2001). Such impacts are to a great extent the result of cattle ranching and agriculture (Dunn 2004; INPE 2009) (www.obt.inpe.br/prodes). Furthermore, most of the agriculture brought with it the extensive use of agrochemicals for pest control, further contributing to the loss of species diversity (Delabie 1990; Ferraz, Russell, Stouffer, Bierregaard, Pimm et al. 2003; Gascon, Williamson & da Fonseca 2000; Perfecto, Mas, Dietsch & Vandermeer 2003; Philpott, Bichier, Rice & Greenberg 2008b). However, despite overwhelming evidence of negative environmental consequences due to deforestation, habitat fragmentation (Broadbent, Asner, Keller, Knapp, Oliveira et al. 2008; Laurance & Bierregaard 1997; Laurance & Curran 2008; Laurance & Peres 2006); simplification of agroecosystems (Dunn 2004; Perfecto et al. 2003; Philpott, Perfecto, Vandermeer & Uno 2009a; Philpott, Soong, Lowenstein, Pulido, Lopez et al. 2009b), and use of pesticides in agriculture (Matlock & de la Cruz 2003; Witt & Samways 2004)**,** new areas are still cleared and new agricultural systems continue to be introduced in the region (Hayashi, Souza Jr & Pereira 2009; INPE 2009) . Consequently, the native vegetation of the Amazon is increasingly intermixed with agricultural systems both small scale (family-run agriculture) and large scale (commercial monocultures), further aggravating the negative impacts on biodiversity (Fearnside 2001).

However, not all agriculture is the same with regards to biodiversity. It has now been well-established that some agroecosystems can sustain elevated biodiversity depending on the type and the intensity of the management used (Delabie, Jahyny, Nascimento, Mariano, Lacau et al. 2007; Harvey & Villalobos 2007; Hernandez-Ruiz & Castano-Meneses 2006; Moguel & Toledo 1999; Perfecto et al. 2003; Philpott, Perfecto, Armbrecht & Parr 2010; Philpott et al. 2009a). Yet other forms of agriculture, specifically the more industrial forms, can have extensive impacts on biological communities (Donald 2004; Hole, Perkins, Wilson, Alexander, Grice et al. 2005; Mader, Fliessbach, Dubois, Gunst, Fried et al. 2002; Reidsma & Ewert 2006; Tilman, Cassman, Matson, Naylor & Polasky 2002; Tilman, Polasky & Lehman 2005).

Since not all types of agriculture have the same effects on biodiversity (Chappell & LaValle 2009; Perfecto, Vandermeer & Wright 2009; Vandermeer & Perfecto 2007), the advancing agricultural frontier and the inevitably related fragmentation of natural habitats call for increased efforts to understand the dynamics of biodiversity within agricultural systems. Many studies have investigated the mechanisms of species loss due to agriculture, both direct (Armbrecht, Rivera & Perfecto 2005; Perfecto 1990; Perfecto, Vandermeer, Hanson & Cartin 1997) and indirect (Ferraz et al. 2003; Laurance et al. 1997; Laurance et al. 2008) but few have focused on the details of which specific environmental variables act as driving forces of biodiversity maintenance within agricultural systems (Cerda, Palacios & Retana 2009; Fayle, Turner, Snaddon, Chey, Chung et al. 2010; Perfecto & Vandermeer 1996; Philpott, Arendt, Armbrecht, Bichier, Diestch et al. 2008a).

For example, studies of coffee and cacao agroforestry systems have found that the canopy layer (i.e. shade level) plays an important role in maintaining ant biodiversity

(Perfecto et al. 2003). On the other hand, few studies have investigated the importance of the herbaceous layer for these same communities (Lubertazzi & Tschinkel 2003), even though the invertebrate fauna may be very different in a system with an herbaceous layer as compared to a system deprived of this vegetation (Blaum, Seymour, Rossmanith, Schwager & Jeltsch 2009; Vargas, Mayhe-Nunes, Queiroz, Souza & Ramos 2007). Furthermore the diversity of plant species in the herbaceous layer is directly related to the quality and quantity of leaf litter. Several studies have shown that leaf litter is the main nesting and foraging space for epigeic ants which nest directly in these environments (Armbrecht & Perfecto 2003; Carvalho & Vasconcelos 1999). Leaf litter also creates microclimate gradients (Mathieu, Grimaldi, Jouquet, Rouland, Lavelle et al. 2009), which are extremely important for these communities (Perfecto et al. 1996) and may change ant species richness as well as composition. Landscape level factors can also prove to be important in maintaining diversity of ants within agroecosystems. For example, it has been shown that the distance to a forest can affect species diversity within coffee plantations, probably due to an abundant source of fertilized queens (Armbrecht et al. 2003; Perfecto & Vandermeer 2002).

Organic agricultural systems can provide positive effects on biodiversity at the medium- and long-terms through interactions within the agroecosystem. Such systems can produce food while establishing an ecological balance that prevents soil degradation, provides pest biocontrol, and reduces the pollution of groundwater through their reduced use of synthetic fertilizers and pesticides (Chappell et al. 2009). Moreover, organic systems can optimize a number of different environmental factors with regards to biodiversity, determined by the local biodiversity, by housing source populations and serving as a migration corridor for the gene pool. The structural features in a diverse system , such as plant species and plant density, can support many different species and optimize nutrient and energy cycling for agricultural production. Further, provision of structures that provide food and shelter for a diverse array of organisms, and the lack of pesticide use, may attract new or re-colonizing species to organic systems (IFOAM 2010).

Recently, orange groves have been introduced into the Amazon region and now form a small but significant piece of the agroecological matrix (Costa & Andrade 2003). Conventional orange orchards in the Amazon are mechanized and use synthetic agrochemicals to control pests and weeds. However, due to the large economic losses caused by soil degradation and increases in pest damage (largely due to resistance and secondary pest outbreaks), some producers are changing to organic management.

The goal of this study was to evaluate ant species richness and composition in orange orchards under organic and conventional management, and to investigate what environmental variables contribute the most to the maintenance of ant diversity in these systems.

MATERIALS AND METHODS

Sampling Areas

The collections were made during the rainy season in February 2008, in the districts of Capitão Poço (01° 44' 56.5" S; 47° 03' 33.7" W) and Garrafão do Norte (01° 56' 12.3" S; 47° 02' 46.2" W) in the northeastern region of the State of Pará, Brazil. Both districts are located at 73 meters above sea level, with an average daily temperature of 25.7° C, and average monthly rainfall of 119.8 mm (SECTAM 2009) (http://www.sectam.pa.gov.br/).

The territorial extent of these districts is 2900 km^2 and 1600 km^2, respectively. Capitão Poço has 9766 ha used for perennial cultures, of which 8610 ha are occupied by orange orchards (IBGE 2006). In Garrafão do Norte, about 1221 ha are occupied by perennial cultures, 500 ha of which are occupied by orange orchards cultivated both organically and conventionally. The northeast of the state of Pará is a mosaic of secondary forests of different ages with pasture systems and small scale agriculture (Davidson, Sa, Carvalho, Figueiredo, Kato et al. 2008).

Methodology of Ant Sampling

Ants were sampled in 20-year-old orange orchards (*Citrus sinensis,* Pêra Rio variety) with two kinds of management: 1) organic (OrgO) and 2) conventional (Co). Three orchards of each management type were selected with minimum distance of 1500 meters between them and an average size of 30 ha. The spacing between rows was six meters, and within the orange tree line was three meters. All organic orchards had been managed as conventional prior to the conversion to organic, which occurred eight years prior. The change to organic management also included the planting of two species of trees **(***Khaya ivorensis* and *Swietenia macrophylla*), scheduled to be cut after at least 15 years. These trees are eight years old, around ten meters tall, and eighteen centimeters in diameter; *S. macrophylla* has extra-floral nectaries (Gouvea, Dornelas & Martinelli 2008).

Ants were collected between 9 and 11 am, with pitfall traps with a diameter of 18 cm and a height of 10.5 cm. In each collection area, 20 traps were installed 15 m apart and arranged in four rows of five pitfalls. Each trap was filled with a solution containing salt and a neutral detergent. The salt was added to preserve the ants, and the detergent was used to break the surface tension of the liquid, reducing the possibility of the specimens escaping. The traps were placed between rows in the orange orchards and left open for 48 hours. We placed pitfall traps between the rows to avoid the dominance of any colony that may be nesting on or below the orange trees. The material collected was transferred to jars with 70% ethanol and taken to the Pheromones and Insect Behavior Lab of the Federal University of Viçosa, in Viçosa in the state of Minas Gerais, Brazil. The material was then sorted, mounted and identified to species or morphospecies based on literature (Bolton 2006; Hölldobler & Wilson 1990) and in comparison with specimens of the Formicidae Collection of the Museum of Zoology of the University of São Paulo.

Methods for Sampling Explanatory Variables

In addition to the type of management (OrgO and CO), the other variables measured were: plant species richness (PS), plant density (DP), average plant height (PH) of the herbaceous layer (up to 1.50 m), distance from the closest forest in meters (Dist), and quantity of leaf litter in grams (Bio). Only the herbaceous layer was considered for all vegetation variables, and neither the orange plants nor the layer above to the herbaceous layer were taken into account. The vegetation variables were collected in a 1-meter radius from the approximate center of the collecting pitfall trap (+/- 9 cm).

The richness and density of the herbaceous plants were determined by collecting, counting, and identifying to morphospecies all plant species within the collecting sites (1 m radius around the pitfall trap). Average plant height was estimated by measuring the heights of up to 10 plants within the collecting sites and taking the average. Leaf litter was estimated by throwing a 0.2 m X 0.2 m square frame within a 2 m radius from the center of the pitfall trap and collecting all litter material within the square. The material was then packed in plastic bags, labeled and transported to the laboratory for dehydration. Then, the leaf litter was quantified in grams (g) by weighing the dry material. Finally, the distance from the forest was determined by measuring the distance between the center of the pitfall trap and the closest forest.

The orange orchards were located within a matrix in flat areas that made it easy to observe the surrounding forest, facilitating the measurement of distances between sampling points and the edge of the forest fragment. At points far away from the Forest, we used a Topcon Pulse Total Station GPT 3000. However, since the region had high relative air humidity due to the rainy season, the marginal measurement error of the Topcon was larger than the range of ant communities. We therefore used this equipment only to determine the closest forest fragment. Afterward, we used a measuring tape to measure the distance between the pitfalls and the edge of the forest fragment.

*Systems**:*** The orange orchards under organic management are fertilized once a year with cattle manure and phosphate (both accepted by organic protocol). Fertilizations are made with a low-compaction tractor. Vegetation is allowed to grow within and between rows for the production of biomass and for soil protection, and is harvested through two manual cuttings a year. The orchards with conventional management receive limestone applications and are fertilized with cattle manure and a conventional (synthetic) fertilizer. The management of the vegetation is done with a mechanical cutter (twice a year) and application of herbicides. In addition, insecticides are used for pest control. The harvest period is between October and December, but in the conventional orchards there is an additional harvest in April consisting of the collection of fruits that were left over from the previous season.

Statistical Analyses

Diversity of Ant Species

Differences in ant species richness as a function of management system were examined using smoothed species accumulation curves with 95% confidence intervals constructed with Estimate S (Colwell & Coddington 1994). Also, shared and unshared species analyses were employed to determine the similarity within organic orange orchard plots and conventional orange orchard plots, and between organic and conventional orange orchards. All these analyses were done using Estimate S (Chao, Chazdon, Colwell & Shen 2005; Colwell et al. 1994).

Relationship between Ant Species Richness and Explanatory Variables

In order to investigate the effect of the explanatory variables on observed ant species richness, a complete model was created with all the variables and double interactions according to the following equation:

Ant species richness ~ System+plot+PS+DP+PH+Dist+Bio+Double interactions

Where: System = type of management used (OrgO and CO); PS = the plant species richness in the herb layer; DP = the density of plants in the herb layer; PH = the height of plants in the herb layer; Dist = the distance between the collecting site and the closest forest; Bio = Leaf litter. The variable *system* was added to the model to differentiate the types of management, OrgO and CO.

Separate analyses were also conducted for each type of management, in which the variable *system* was removed from the model. The minimum adequate model (MAM) was selected via *stepwise* (stepAIC), because the method allows the introduction and removal of variables in different configurations of the model until it reaches the most parsimonious minimum adequate model (Venables & Ripley 2002). However, in the data explanatory phase, an explanatory statistical analysis was conduced to guide us to towards a final model, as suggested by Whittingham and colleagues (2006). Model selection via Akaike's Information Criterion (AIC) was used, as proposed by Whittingham et al. (2006), but with alteration of the parameter k of the function stepAIC from 2.71 ($p = 0.1$) to 3.84 ($p = 0.05$), in order to increase the strictness of the criterion to add and remove variables from the model. The estimation method utilized was a Generalized Linear Model with a Poisson error distribution because of our discrete data (Crawley 2009) and this was the error distribution which best fit the data set. The selected variables in the MAM were submitted to a *Deviance* analysis and only were maintained in the model if they were significant at 5%. After the selection of the MAM, the residues were analyzed to verify its adequacy and the distribution of the errors used. All analyses were done using R and Estimate S (Colwell et al. 1994; Hornik 2009; R Development Core Team 2010).

RESULTS

Ant fauna

Overall, eighty-five species of ants were collected, distributed in 7 sub-families: Myrmicinae, with 44 species in 10 genera; Formicinae, with 18 species in 3 genera; Dolichoderinae, with 8 species in 4 genera; Pseudomyrmecinae, with 7 species in 1 genus;

Ectatomminae, with 3 species in 2 genera; Ponerinae, with 3 species in 3 genera; and Ecitoninae, with 2 species in 2 genera.

In the orchards under organic management (OrgO) seventy species of ants were collected, nineteen of which were exclusive to that management type. In the conventional orchards (CO), sixty-six species were sampled, fifteen of which were exclusive to the CO type (Table 1). The genus with the highest richness was *Pheidole*, with 13 species, followed by *Camponotus*, with 11 species. The species found with highest frequency in the organic orchards were *Crematogaster quadriformis* (29 occurrences), *Ectatomma brunneum* (28 occurrences), *Solenopsis geminata* (25 occurrences) and *Pheidole* (gr. *fallax*) sp36 (23 occurrences). In the conventional orchards, the most frequent species were *Solenopsis geminata* (50 occurrences), *Ectatomma brunneum* (43 occurrences), *Pheidole fallax* (43 occurrences) and *Wasmannia auropunctata* (37 occurrences). Although some of the orange trees hosted colonies from genera like *Dolichoderus*, *Azteca* and *Crematogaster*, we did not consider these to be solely arboreal due their presence in litter layer, and the presence of multiple nests on different trees for some species.

The reduced species richness and the presence of army ants (and other predator species, such as *Dinoponera gigantea*) caused some concern. The sampled region is within their distribution range, and thus it would be expected that they would be collected in these areas; this is a possible concern due their important role in the control of different species such as *Solenopsis* and *Atta*.

Smoothed species sample curves show no significant differences in species richness between the organic and conventional orchards (Figure 1). When species composition was considered, more than 44% of species were unshared among organic orchard plots and

42.5% of species were unshared among conventional orchard plots. The mean shared species between organic and conventional orchards was only 42.5%.

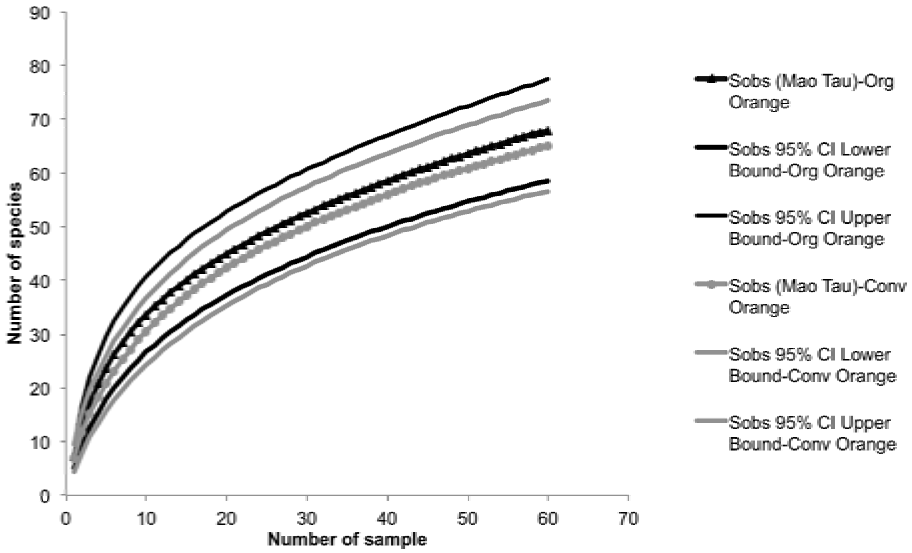


Figure 1: Smoothed species sample curves for organic and conventional orange orchards with 95% confidence intervals.

Pest Species in Orange Orchards

Four ant species, *Solenopsis geminata*, *S. saevissima*, *S. globularia* and *W. auropunctata* are known to cause serious problems in orange orchards due to their aggressiveness toward humans (Fowler, Bernardi, Delabie, Forti & Pereiradasilva 1990). In the orange orchards, *Solenopsis* species were frequent, with two species being common (> 20% frequency) in both orchards (Figure 2). The conventional orange orchards had greater abundance and frequency of both *S. geminata* and *W. auropunctata* species, reaching twice the frequency seen in the organic orchards (Figure 2). Moreover, an analysis of variance in this species group showed marginally significant differences in frequency between organic and conventional orchard (p= 0.058).

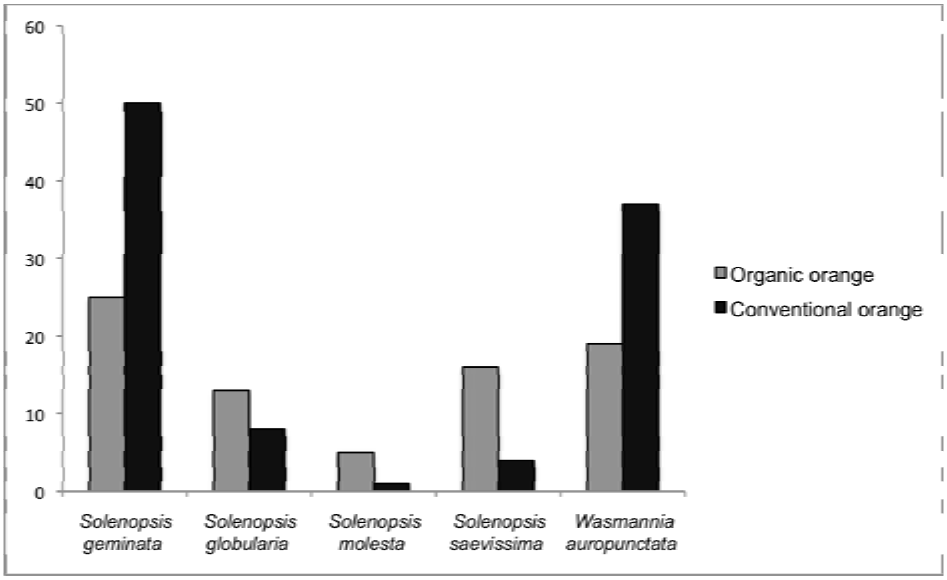


Figure 2: Frequency of ant species that cause problems in organic and conventional orange orchards in Brazil.

Ant Species Richness and Explanatory Variables

When a One-Way ANOVA was used to compare management systems (OrgO and CO) without considering environmental variables, the levels of ant species richness did not show a significant difference ($p>0.05$). However, when environmental variables were considered within and between management types, the results were quite different. When only the organic management (Org) was considered, leaf litter (Bio) explained the variability in ant species richness ($p_{Bio}<0.005$), with higher species richness in areas with high leaf litter. Secondarily, the interaction between plant height (PH) and leaf litter (Bio) showed positive effects on ant species richness ($p_{PH*Bio}= 0.043$), and the interaction between the distance from the forest and the leaf litter was marginally significant ($p_{Dist*Bio}=0.094$) (Table 2).

Neither plant species richness (PS), density of plants (DP), height of plants (PH**)** nor distance from the forest (Dist**)** had significant effects on ant species richness in the organic orchards ($p_{PS}= 0.218$, $p_{DP}= 0.732$, $p_{PH}= 0.854$, $p_{Dist}= 0.682$).

Considering only the conventional management system, plant species richness (PS) in the herbaceous layer had a significant effect on ant species richness ($p_{ps}= 0.009$). In addition, the following interactions were significant: plant species richness (PS) and distance from the forest (Dist); plant species richness (PS) and leaf litter (Bio) ($p_{PS*Dist}= 0.05$; $p_{PS*Bio}=0.009$) (Table 2). The importance of each variable and the interactions between them changed in relation to the distance from the forest. In other words, the variables had different effects and magnitude on ant species richness depending on distance from the forests (Figure 3).

When considered individually, the leaf litter, the height of the plants and the distance from the forest did not present significant effects on the ant species richness for the conventional orchards (Table 2).

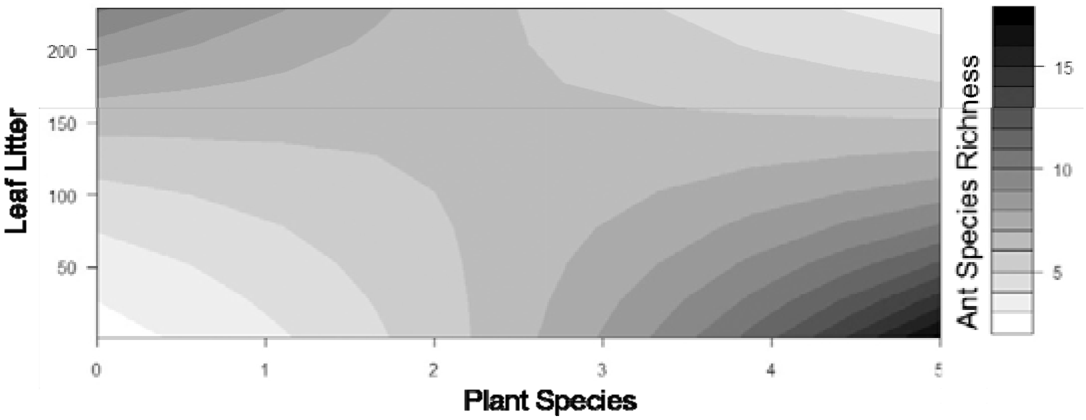
When both types of management were grouped together (OrgO and CO), the environmental variables and the interactions had different behavior and/or magnitude. In general, however, the importance of the independent variables was retained when the systems were grouped. In both systems, the sites with greater plant species richness had greater ant species richness (p_{SP}=0.016). However, some variables became more important when combined with other variables. For example, quantity of leaf leaf litter, plant height, and distance from the forest, which showed no significant effect on ant species richness when considered independently from the other variables (p_{Bio}= 0.388; p_{PH}= 0.954; p_{Dist}= 0.524), showed significant interactions with other variables (p_{DP*PH}= 0.002; $p_{Dist*Bio}$= 0.003). Even when the two types of management were analyzed together, the quantity of leaf litter in the conventional system was strongly positively related to ant species richness ($p_{Systema\_CO*Bio}$= 0.006). The interaction between plant height and distance from the forest showed marginal effects on ant species richness in both types of orchards ($p_{PH*DIst}$= 0.072). On the other hand, the direction and intensity of the effect of plant height and leaf litter depended on the distance from the forest ($p_{PH*Dist}$= 0.072; $p_{Dist*Bio}$= 0.003). When the orchards were close to the forest (17.1 m), plant species richness was the variable that determined the ant species richness. When the orchards were at medium distances (412.52 m), plant species richness lost some of its importance, while leaf litter gained more importance in determining ant species richness. However, when the orchards were at long distances from the forest (910 m), plant species richness lost its importance and leaf litter determined ant species richness (Figure 3 and Figure 4).

On the other hand, when considering the absolute value of leaf litter and the mean of the distance to the Forest (at distances <275 m) in a linear model of the relationship between leaf litter and ant species richness, the significant effects on ant species richness was lost when the orchards was closest (17.1 m). The effect was likely diluted as it moved away from the forest edge to the orchards center (Figure 5). In the same way, at medium and far distances from Forest (between 275-560 m and >560 m), leaf litter appeared important to support high levels of ants species richness in conventional orange orchards, but had no effect in organic orange orchards (Figure 5). Nevertheless, when the same analysis as above for short distances from the forest (linear and with just two axes) was conducted with plant species richness, the effects for both systems (Conventional an Organic) was insignificant or negative effect with regards to ant species richness (Figure 6). Thus it appears that there is a synergistic effect among different variables, clearly displaying the importance of interactions of environmental variables between and within orange orchard management types (Figures 3, 4, 5, 6).

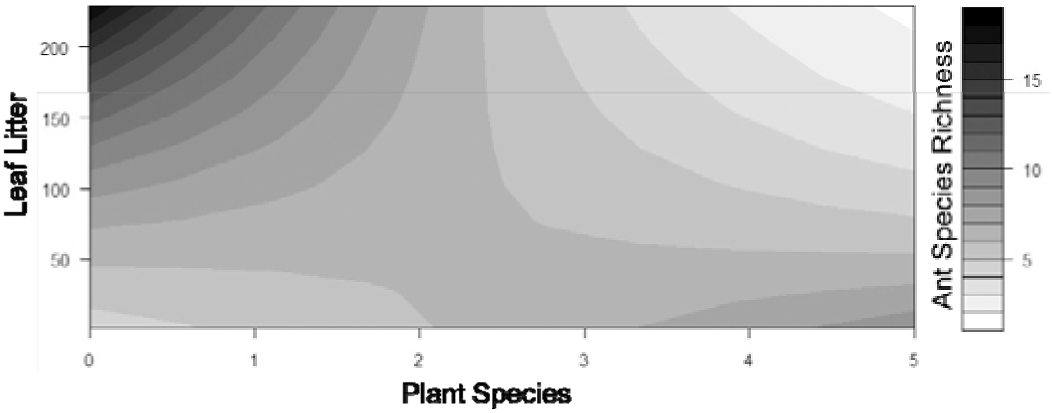
The density of plants had an important role in determining ant species richness in the orchards but, when grouped with other variables such as distance from the forest and quantity of leaf litter, these two variables prevailed in relation to the density of the plants (Figure 4). For example, leaf litter was the variable that showed the strongest effect on ant species richness in sites without plant cover, as well as in the sites with high plant density.

Looking the positive effect of many environmental variables, as well as of the importance of assorted positive interactions between different environmental variables, we can deduce that the heterogeneity in both orange management systems can be the determining factor for biodiversity (Table 2 and Figure 3). Plant species richness, leaf litter, plant height and plant density had stronger significant effects when they were modeled together than alone. So, the synergism between them can be interpreted as emphasizing the broader importance of heterogeneity within orange orchards in supporting high level of ant species richness (Table 2).

Distance from the Forest (17.1 m)



Distance from the Forest (412.52 m)



Distance from the Forest (910 m)

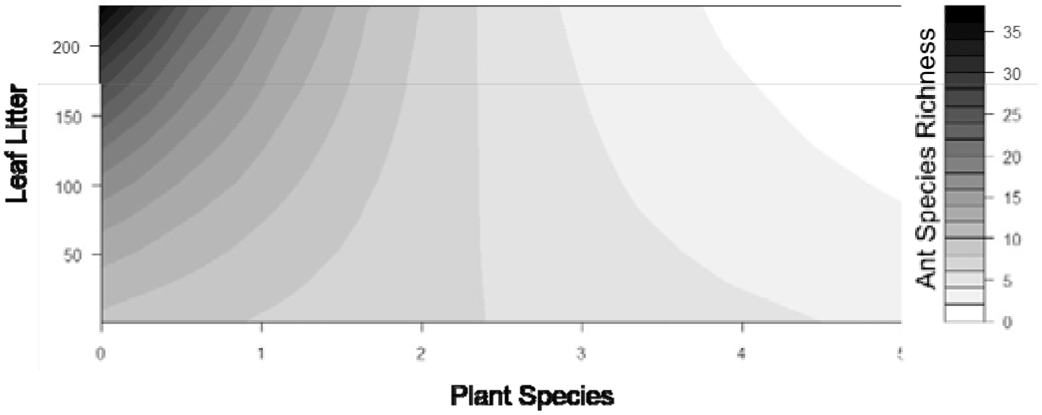
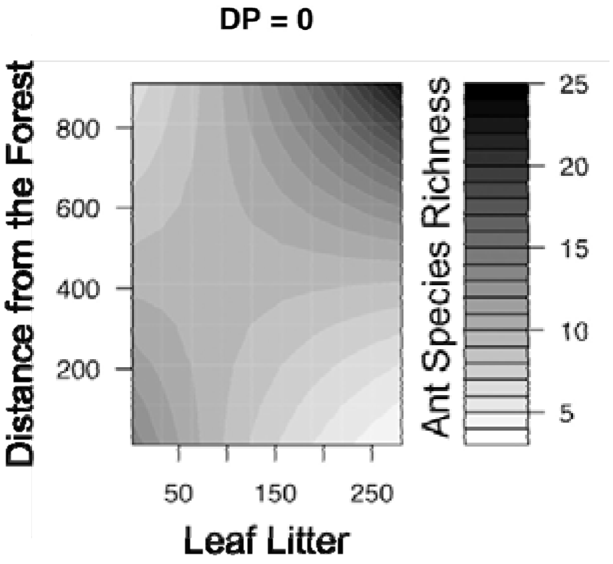
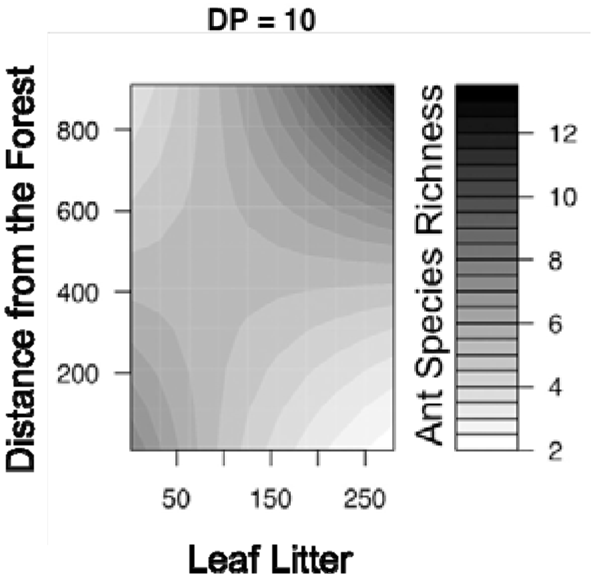


Figure 3: Effect of plant species richness and leaf litter on ant species richness in orange orchards (both conventional and organic) according to the distance from the forest. This

shows ant species richness predicted by our minimum adequate model under a range of distances from the forest for plant species richness and leaf litter. The points on white color represent areas without sampling of that variable.





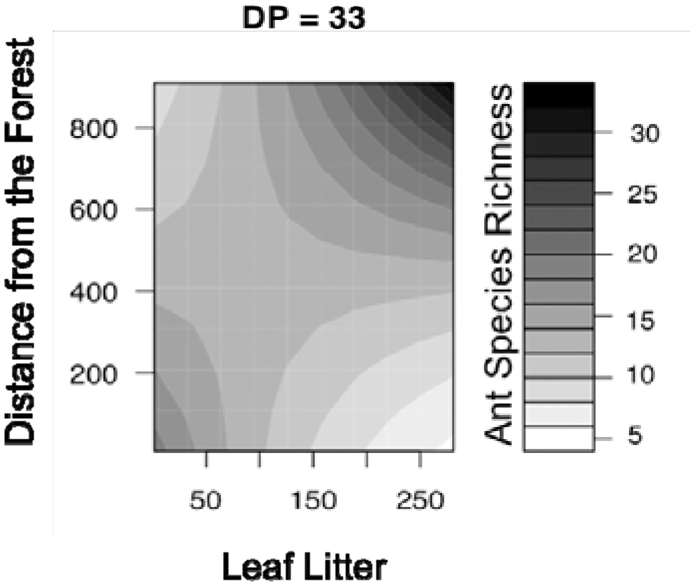


Figure 4: Ant species richness in three categories of density of plant in relation to the distance from the forest and quantity of leaf litter. This shows ant species richness predicted by our minimum adequate model under a range distances from the forest. The points on white color represent areas without sampling of that variable.

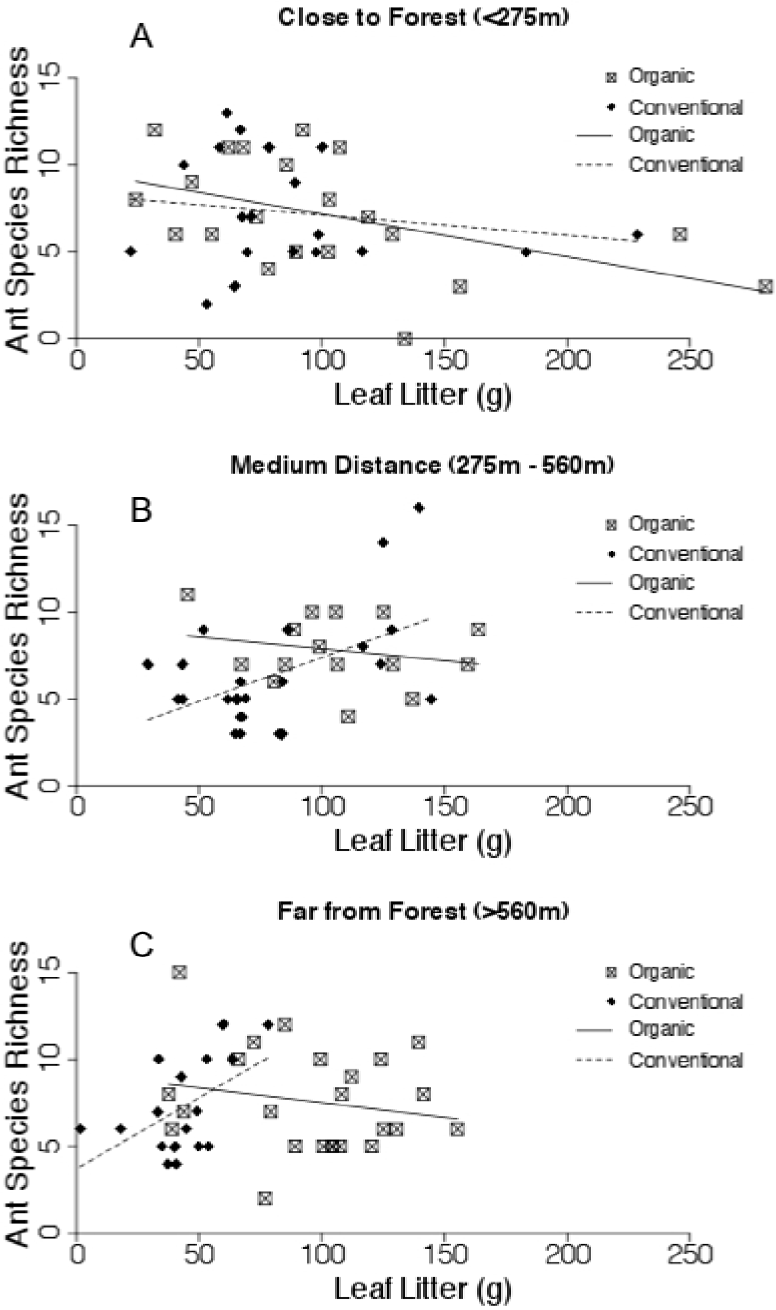


Figure 5: Effect of leaf litter on ant species richness in orange orchards (both conventional and organic) according to the distance from the forest. At this graphic is presented a linear relationship between ant species richness and leaf litter in absolute value of each variable.

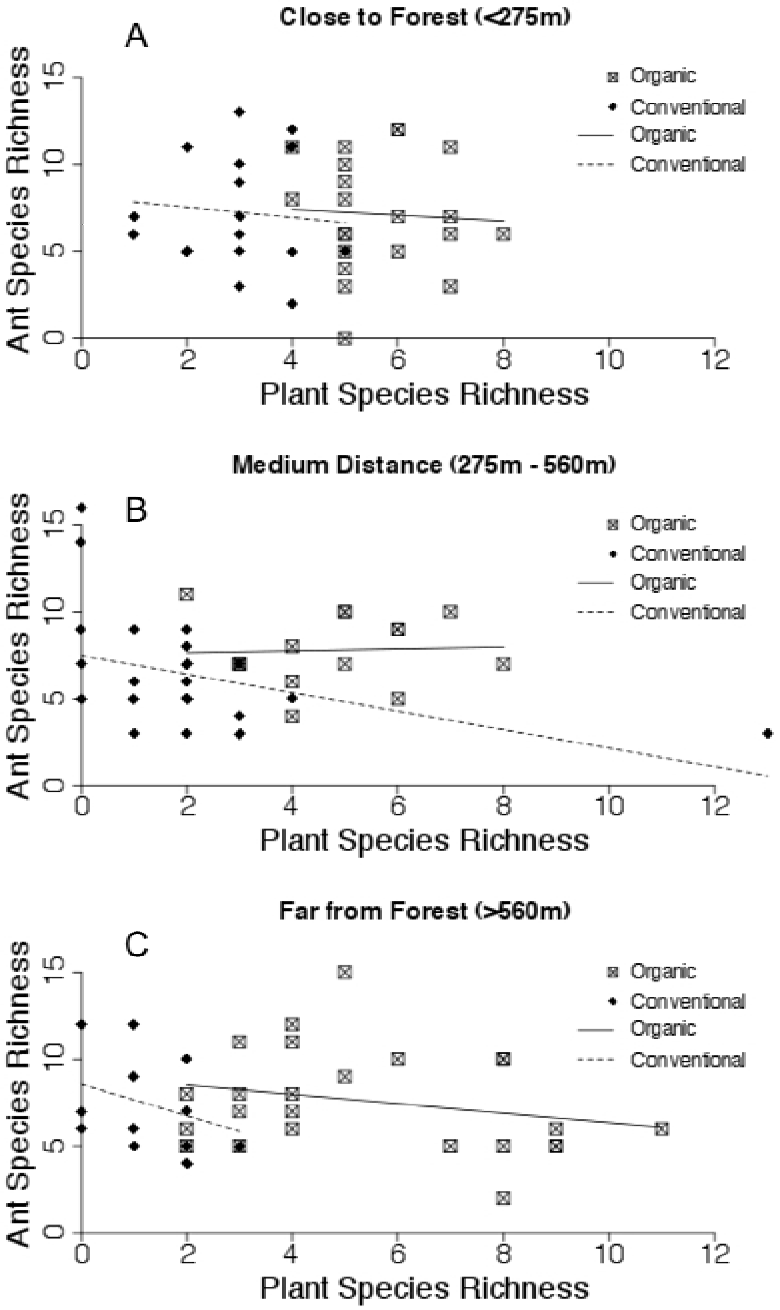


Figure 6: Effect of plant species richness on ant species richness in orange orchards (both conventional and organic) according to the distance from the forest. At this graphic is presented a linear relationship between ant species richness and plant species richness in absolute value of each variable.

DISCUSSION

A total of 85 species of ants were collected in the 6 orange orchards sampled, a remarkably high diversity considering that pitfall traps were kept open for a single period of 48 hours in each orchard. However, this is not as high as the number of species that were collected in traditional agroforestry systems in the same region (Dos Santos, in preparation). Despite the differences in management, we found no significant differences in ant species richness between organic and conventional orchards. The percent of unshared species was similar within organic and conventional orchard and between the two management systems (44% and 42.5% respectively). In spite of the lack of significant differences of species richness between organic and conventional orchards, when examining the vegetation and landscape variables measured, we found many differences that appear to explain the levels of ant species richness between the two types of orchards.

Organic Orange Orchards

The quantity of leaf litter was the variable that most contributed to the increase of ant species richness in the orange orchards under organic management, most likely because this is the fundamental resource for epigeal ant communities, representing its main foraging and/or nesting area (Armbrecht, Perfecto & Silverman 2006; Carvalho et al. 1999). It is also possible that organic orchards have a more diverse leaf litter composition because, in addition to the oranges, they plant two timber tree species (Kato, unpublished data); Armbrecht and colleagues (2006) showed that ant species richness increased with increased diversity of twig species in the leaf litter. On the other hand, the importance of herbaceous plant species richness, plant density and plant height can be indirect, since all of these variables are fundamental for the maintenance of the quantity of leaf litter. This is supported by the significant positive interaction found between plant height and leaf litter (p_{PH*Bio}= 0.043), which can be explained by the fact that taller plants may have greater productivity of vegetative material and can increase the quantity of leaf litter (Vieira, de Almeida, Davidson, Stone, de Carvalho et al. 2003).

Likewise, other variables, like plant density, that show less of an affect when examined independently, can also have an indirect effect by increasing leaf litter. Since organic orchards are not as intensively managed as conventional ones, their herbaceous vegetation gets less disturbed (in intensity and frequency) than in the conventional orchards. All of this contributes to higher quantity and perhaps quality of litter in organic orchards.

Conventional Orange Orchards

In the orange orchards under conventional management, the most important variables for ant species richness were herbaceous plant species richness and plant density, as well as several interactions with other variables.

Some variables, like plant height and distance from the forest, were not significant when considered independently. However, in combination with other variables, the relation was significant. In other words, there was some interdependency between the variables. Furthermore, several interaction terms were highly significant in explaining ant species richness in these orchards. The fact that more variables and interactions explained ant richness in conventional than in organic orchards may have to do with the intensity and frequency of disturbance of the soil and leaf litter biomass (Bestelmeyer & Wiens 1996; de Bruyn 1999; Kaspari 1996) and herbaceous layers in these orchards. Agricultural practices in the CO orchards, such as grass cutting and application of herbicides, alter the herb layer and also change plant species composition, plant density and leaf litter. This variability in vegetation and litter structure may directly affect ant communities (Armbrecht et al. 2006; Philpott et al. 2009b). In addition, unlike organic orchards, conventional orchards have no other species of trees in the canopy. All of these management factors combine to make the herbaceous and litter layers in conventional orchards more variable. Consequently, the resources may have been more variable, making a larger number of environmental variables important for these communities of ants.

Organic and Conventional Orchard Together

Since no difference in ant species richness was detected between conventional and organic orchards, we examined the environmental variables in a model that included orchard type as a fixed variable. When the two systems were grouped together, the patterns of significance of the environmental variables and interactions were the same, but with different strengths (Figures 2 and 3).

In general, and contrary to what has been found in other studies, ant species richness increased with distance to the forest (Armbrecht et al. 2003; Hernandez-Ruiz et al. 2006; Philpott et al. 2009b). This is an enigmatic result and one for which we present two potential explanations in a local scale. First, it is possible that when the orange orchard is close to the forest, the majority of the species prefer the forest instead of the orchards because the forest has better microhabitats for nesting, more diversity of resources and better conditions to maintain their colonies (Armbrecht et al. 2003; Armbrecht et al. 2006; Philpott et al. 2009b). Second, a reduced number of species that have their nests in the forest come out to forage in the orange orchards but the capacity of dispersion of the workers is not exceeded by the distance and it would justify the greater importance of the plant species on the ant species richness because some species may continue perceiving the border of the orchard as a continuation of the forest. This might have been the case for *Azteca* sp30 and *Cephalotes atratus*, which are strictly arboreal species and therefore more dependent on trees. Whatever the reason is for an increase in ant species diversity with distance from the forest, it highlights the importance of managed systems based on trees for landscapes that are highly deforested and/or fragmented, like the one where this study was conducted.

The sampled orchards are located in the region called the “Deforestation Arch”, the region of the Amazon basin with the highest rates of both deforestation and immigration (Ferraz et al. 2003; Ferraz, Capao & Vettorazzi 2006; Laurance, Goosem & Laurance 2009; Michalski, Peres & Lake 2008; Skole & Tucker 1993). It is where the more intensive agricultural systems are concentrated (INPE 2009). This region is also the oldest agricultural frontier of the Amazon, and therefore has been more degraded, and for a longer period of time, than the rest of the Brazilian Amazon. Recent evidence suggests that these systems are starting to collapse (Secretaria Municipal de Agricultura, unpublished data).

Intensive agricultural systems have caused significant losses of biodiversity throughout the world (Delabie et al. 2007; Donald 2004; Matson, Parton, Power & Swift 1997; Perfecto et al. 2003; Philpott et al. 2008a; Philpott et al. 2008b; Reidsma et al. 2006; Tilman et al. 2002). It is well documented that forests and less intensive agricultural systems (traditional, organic, ecological, low input, etc) contain more biodiversity than agroecosystems with more intensive management (Crist 2009; Perfecto et al. 2003; Philpott et al. 2008b; Reidsma et al. 2006; Roberts, Cooper & Petit 2000). However in our study, conventional orchards and organic orchards did not differ in terms of ant species richness. Likewise, some species indicative of better conserved areas, such as *Eciton burchellii* and *Dinoponera gigantea*, were missing from both types of orchards. This is indicative of the biological degradation in the region as a whole, but also reflects the fact that orange orchards, even the ones that are managed organically, are relatively intensive systems that significantly alter the natural environmental conditions of the soil and litter layer where most of the epigeal ant community lives. Since the organic orchards had been managed intensively for 20 years before the relatively recent conversion to organic eight years ago, the lack of difference in ant richness between these two systems could also be a reflection of the long time that it takes to recover biodiversity after intensive agriculture (Mathieu et al. 2009) This could be especially true in landscapes that are dominated by intensive agriculture and lack forested habitats that serve as sources of biodiversity (Perfecto et al. 2009).

Army ants are important predators in the neotropical regions (Powell & Baker 2008) and promote the control of several species of invertebrates, mainly other species of ants, such as *Atta* and *Solenopsis* (Perfecto 1992; Powell et al. 2008; Powell & Clark 2004). Furthermore, most of the army ant species are sensitive to environments that are open, with few resources, dry and with elevated temperatures (Roberts et al. 2000). Roberts et al. (2000) showed that more isolated and less shaded coffee plantations had half of the potential nesting sites for communities of army ants when compared to the forests and to the shaded coffee. The conventional orange orchards are fairly open environments with elevated temperature and few or almost no nesting sites for these species of ants, probably accounting for their absence. The absence of these and other species of predatory ants might be responsible for the dominance of *S. geminata* and *W. auropunctata* in these orchards, since both have the ability to dominate other species and colonize environments that are intensely disturbed (Foucaud, Orivel, Fournier, Delabie, Loiseau et al. 2009; King, Tschinkel & Ross 2009; Orivel, Grangier, Foucaud, Le Breton, Andres et al. 2009).

CONCLUSIONS

Species richness was similar in organic and conventional orchards, a result at odds with other studies (Letourneau & Bothwell 2008). It is possible that the conversion to organic, just eight years ago, has not allowed enough time for the biodiversity of these orchards to increase, as has been shown for other systems (Mathieu et al. 2009). Nonetheless there were many differences in species composition and species overlap between the two types of orchards. Likewise many differences were found in the environmental variables that explain ant species richness in the two types of orchards. The leaf litter was the most important environmental variable related to ant species richness in orange orchards under organic management, while herbaceous plant species richness and plant density were the most important variables in conventional orchards.

When both systems were considered together, environmental variables were interdependent and the sites with more plant species, greater quantities of leaf litter, and higher plant density, had higher ant species richness in both kinds of orchards. This relation, however, is altered by the distance between the orange orchards and the forest.

In this work, treating the environmental variables as measurements of the environmental structure showed that sites with more diversity and plant biomass may provide more resources to the ant community. Moreover, the increase in ant pest species in the conventional orchards suggest that it would be beneficial for the farmers in the region to change from conventionally managed orchards to less intensive management with fewer negative impacts, like organic. Furthermore, the results of this study suggest that in landscapes that are dominated by intensive annual cropping systems and pastures, managed systems that incorporate an arboreal component, like orange orchards, could be important for the conservation of ant diversity.

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Table 1: Ant list species and frequency in orange orchards.

Ant species	**Management System** **Organic orange**	 **Conven orange**
Dolichoderinae		
Azteca sp30	1	0
Dolichoderus bispinosus	1	0
Dolichoderus diversus	0	1
Dolichoderus quadridenticulatus	9	7
Dorymyrmex (px) *jheringi*	1	5
Dorymyrmex brunneus	8	18
Dorymyrmex spurius	1	1
Tapinoma melanocephalum	0	1
Formicinae		
Brachymyrmex pictus	1	0
Brachymyrmex sp20	10	6
Brachymyrmex sp21	16	3
Brachymyrmex sp22	2	1
Camponotus atriceps	0	2
Camponotus blandus	15	15
Camponotus cingulatus	10	14
Camponotus crassus	12	12
Camponotus renggeri	4	3
Camponotus sexguttatus	0	1
Camponotus sp20	3	0
Camponotus sp21	10	10
Camponotus sp26	1	0
Camponotus sp27	1	1
Camponotus sp4	1	0
Paratrechina fulva	4	2
Paratrechina longicornis	1	3
Paratrechina sp20	6	1
Myrmicinae		
Cardiocondyla nuda	1	0
Cardiocondyla sp2	3	1
Cephalotes atratus	2	1

Cephalotes pusillus	0	1
Crematogaster sp22	1	1
Crematogaster sp31	1	0
Crematogaster sp32	0	1
Crematogaster abstinens	22	15
Crematogaster crinosa	0	1
Crematogaster quadriformis	29	31
Crematogaster sp20	3	2
Crematogaster spC	1	0
Cyphomyrmex sp1	2	1
Cyphomyrmex sp11	0	2
Cyphomyrmex sp2	1	1
Cyphomyrmex sp3	0	2
Cyphomyrmex sp4	14	6
Cyphomyrmex sp6	1	2
Cyphomyrmex sp8	0	1
Cyphomyrmex sp9	0	2
Monomorium floricola	2	0
Mycocepurus smithii	1	6
Pheidole (gr. *fallax)* sp36	23	10
Pheidole (gr. *flavens)* sp1	8	6
Pheidole fallax	20	43
Pheidole flavens	0	2
Pheidole gertrudae	4	8
Pheidole sp20	9	3
Pheidole sp24	5	7
Pheidole sp25	23	10
Pheidole sp26	1	6
Pheidole sp27	0	6
Pheidole sp30	2	0
Pheidole sp40	1	1
Pheidole transversostriata	0	1
Solenopsis geminata	25	50
Solenopsis globularia	13	8
Solenopsis molesta	5	1
Solenopsis saevissima	16	4
Strumigenys (px.) *mandibularis*	1	0
Strumigenys eggersi	1	0
Strumigenys grytava	6	1
Wasmannia auropunctata	19	37

Wasmannia sigmoidea	7	3
Ectatomminae		
Ectatomma brunneum	28	43
Ectatomma tuberculatum	2	0
Gnamptogenys rastrata	2	2
Ponerinae		
Hypoponera sp1	1	0
Odontomachus bauri	9	8
Pachycondyla harpax	1	0
Ecitoninae		
Labidus coecus	3	4
Nomamyrmex esenbeckii	1	0
Pseudomyrmecinae		
Pseudomyrmex (gr. *pallidus)* sp1	0	1
Pseudomyrmex (px. *simplex)* sp2	2	1
Pseudomyrmex filiformis	1	0
Pseudomyrmex gracilis	1	1
Pseudomyrmex holmgreni	2	1
Pseudomyrmex tenuis	1	0
Pseudomyrmex termitarius	18	8
Total= 85 species	463	460

Table 2: Environmental variables parameter in organic and conventional orange orchards.

Organic Orange Orchards					
	Df	**Deviance**	**Resid. Df**	**Resid. Dev**	**P(>|Chi|)**
NULL			59	71.161	
Plant species	1	1.5164	58	69.645	0.218164
Density of plants	1	0.1172	57	69.527	0.732045
Plant height	1	0.0336	56	69.494	0.854551
Distance from the forest	1	0.1676	55	69.326	0.682263
Leaf litter	1	8.8914	54	60.435	0.002865 **
Plant species * Density of plants	1	1.2454	53	59.189	0.264437
Plant species * Plant height	1	0.0018	52	59.188	0.965769
Plant species * Distance from the forest	1	0.9114	51	58.276	0.339745
Plant species * Leaf litter	1	0.1103	50	58.166	0.739768
Density of plants * Plant height	1	0.8975	49	57.268	0.343460
Density of plants * Distance from the forest	1	0.1133	48	57.155	0.736458
Density of plants * Leaf litter	1	0.0041	47	57.151	0.948915
Plant height * Distance from the forest	1	0.0899	46	57.061	0.764312
Plant height * Leaf litter	1	4.0906	45	52.971	0.043123 *
Distance from the forest * Leaf litter	1	2.8030	44	50.168	0.094086 .
Conventional Orange Orchards					
NULL			59	77.312	
Plant species	1	6.7048	58	70.607	0.009615**
Distance from the forest	1	0.0001	57	70.607	0.994284
Leaf litter	1	1.8332	56	68.774	0.175754
Plant species * Distance from the forest	1	3.8174	55	64.956	0.050723.
Plant species * Leaf litter	1	6.6955	54	58.261	0.009666**
Organic and Conventional Orange Orchards					
NULL			119	149.89	
System	1	1.4199	118	148.47	0.233427
Plant species	1	5.7995	117	142.67	0.016031*
Density of plants	1	0.2369	116	142.44	0.626426
Plant height	1	0.0032	115	142.43	0.954993
Distance from the forest	1	0.4051	114	142.03	0.524467
Leaf litter	1	0.7437	113	141.28	0.388469
System(CO) * Plant species	1	2.6514	112	138.63	0.103460
System(CO) * Leaf litter	1	7.4015	111	131.23	0.006517**
Plant species * Plant height	1	0.4948	110	130.74	0.481799
Density of plants * Plant height	1	9.4498	109	121.29	0.002112**
Plant height * Distance from the forest	1	3.2186	108	118.07	0.072804.
Distance from the forest * Leaf litter	1	8.6997	107	109.37	0.003183**

GENERAL CONCLUSION

In this thesis I examined the impacts of various land use systems on the diversity, abundance and community structure of the ant community of the Amazon region. I also investigated environmental variables that may contribute to support species richness within these systems. Figure C.1 shows smooth species accumulation curves for the seven systems examined: soybean, pastures, teak plantations, *paricá* plantations, conventional orange orchards, organic orange orchards and agroforestry systems. From these curves, it is obvious that soybean, pastures and teak plantations support a very low diversity of ant species, and that agroforestry systems support the highest ant diversity of all the systems examined. *Paricá* plantations and orange orchards, both conventional and organic, support intermediate levels of ant diversity. The environmental variables that we measured were grouped into, vegetational variables (Plant Species, Plant Density, Plant Height and Leaf Litter Biomass) soil nutrient variables (soil depth (DS) nitrogen (N%), organic matter (MO [g/kg]), pH (water), phosphorous (P[mg/dm^3]), potassium (K[mg/dm^3]), Calcium (Ca [cmol$_c$/dm^3]), the base Calcium-Magnesium (Ca+Mg [cmol$_c$/dm^3]), and Aluminum (Al [cmol$_c$/dm^3]), and one landscape level variable (distance to the nearest forest). Overall, soil variable explained very little of the variation in ant species richness in all the systems. The vegetational variables were highly correlated, for some systems, such as Agroforestry, Soybean and Pasture, but leaf litter biomass was the variable that help explained most of the variation in ant species richness and abundance within most of the systems. Finally, distance to the nearest forest was also an important variable to explain ant species richness especially for Soybean, Pasture and Teak planted-forest.

In what follows I summarize the main findings and conclusions from the 4 papers that emerged from this dissertation.

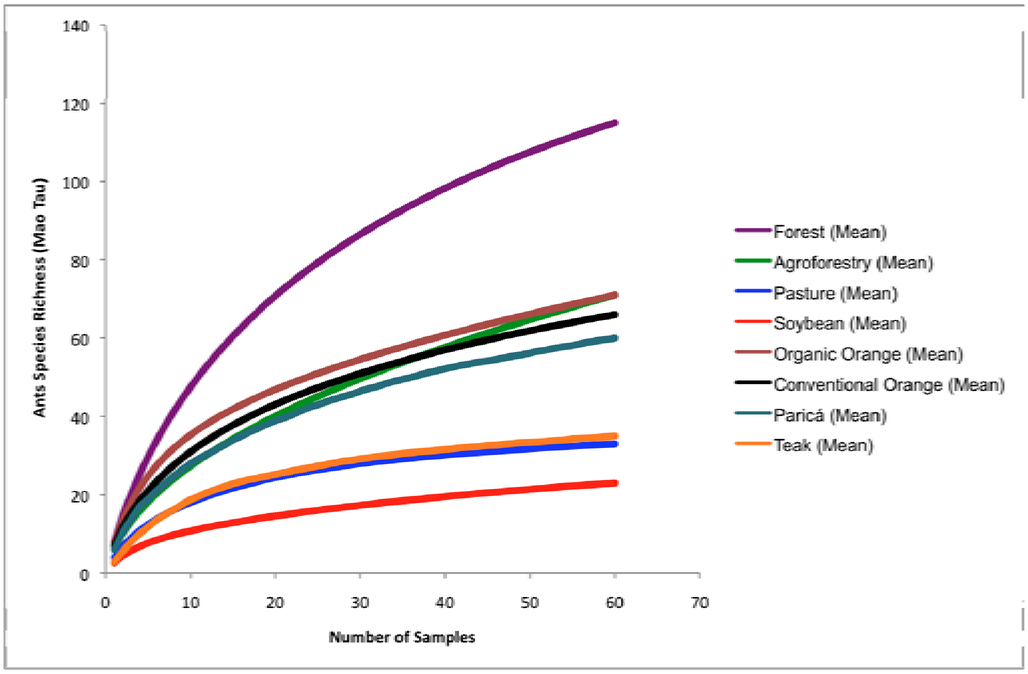

Figure C.1: Ant species accumulation curve (Mao Tau - 95% CI) for Forest, Forest, Agroforestry (SAFS), Pasture and Soybean, Organic Orange and Conventional Orange agroecosystems, and Paricá and Teak planted-forest.

Paper 1: Biodiversity declines following the expansion of soybean and cattle ranching in the Amazon.

In this study we use the ant community as a model group to assess the impacts of soybean, pasture and agroforestry systems on biodiversity. The results demonstrate that soybean plantations and cattle pastures have significantly lower ant species richness than agroforestry systems. Many ant species were found exclusively in the agroforestry systems (Forty-one ant species), compared to just three species unique to pasture and four to soybean, the latter being typical species from disturbed environments. Using a Conversion System Rate Model (CSRM) based on a re-sampling analysis we predicted ant species richness under various scenarios of landscapes composed of different percentages of pasture, soybean and agroforestry. This model predicts that as the landscape is converted from agroforestry systems to soybean or pastures, ant species richness is dramatically lost (Figure 2 at the first paper). The highest ant species richness is found in an all-agroforestry system landscape. With just 20% of agroforestry replaced by either soybean or pasture the ant biodiversity declines dramatically. As pasture is replaced by soybean, as is currently happening in the Amazon region, there is only a small decrease in ant biodiversity because the two agroecosystems have a similar number of species.

The model results suggest that we can recover a lot of the ant species richness by converting some of the pasture and soybean land into agroforestry systems. Therefore, we conclude that agroforestry represents a much better land use system for the protection of biodiversity in the Amazon region.

Paper 2: Soybean expansion and the loss of biodiversity in the Amazon.

This paper focuses on the environmental variables that may help explain the patterns of ant species richness in soybean plantations, cattle pastures and agroforestry systems. We constructed a predictive Bayesian model to evaluate which of the soil and environmental variables (including distance to nearest forest) best explained ant species richness across the three agroecosystems. Many environmental variables had strong effects on ant species richness when examining all systems together. Also we used the parameters from the environmental variables prediction model to investigate how ant species richness would be affected under different conditions for three selected environmental variables - *distance to forest fragment, herbaceous plant density and leaf-littler biomass*. Predictions show that pastures closer to forest fragments had more ant biodiversity than soybean farms, but when both were farther away this difference disappears (Figure 3 at the second paper). On the other hand agroforestry systems had higher ant species richness than soybean or pastures no matter how far from a forest fragment. For low or high plant density agroforestry systems had considerably higher ant species richness than either soybean or pasture, which had roughly the same ant species richness in both low and high plant density. Leaf-litter biomass had the largest effect on ant biodiversity across all systems, doubling the number of ant species observed (Figures 2 and 3 at the second paper). Still in both low and high leaf-litter biomass conditions agroforestry systems had the highest ant biodiversity.

Paper 3: The impact of forest plantations of native and exotic species on ant biodiversity in the Amazon region.

In this paper we evaluated the ant biodiversity in timber plantations planted with *paricá* (*Schizolobium amazonicum*), a native tree, and teak (*Tectona grandis*) an exotic tree, both planted as timber trees in the Amazon region. We found almost double the number of ant species in the native plantations as compared to the exotic plantations (Figure 1 at the third paper). Approximately half of the species pool were species unique to the native plantations (29 species) compared to just four in the exotic plantations. Leaf litter biomass, which is typically correlated with ant diversity, had a positive effect on ant species richness in the native plantations, but no effect in the exotic plantations (Figure 3 at the third paper).

This is probably because the teak plantations produce a very homogeneous layer of leaf litter.

Paper 4: Effect of Organic and Conventional Orange Orchard Management on Ant Diversity in the Amazon.

In this paper our goal was to evaluate ant species richness and composition in orange orchards under organic and conventional management and to investigate what environmental variables influence ant diversity within these systems. We found a total of 85 ant species, 69 in organic and 65 in conventional orchards. Smoothed species/sample curves show no significant differences on species richness between systems, nor in terms of percent of shared species within the two types of orchards, nonetheless there was less than 50% species overlap between conventional and organic orchards. Furthermore, there were significantly more “pest” species in the conventional orchards as compared to the organic (Figure 4 at the forth paper). Under conventional management, herbaceous plant species richness and herbaceous plant density were highly correlated with ant species richness, while in organic orchards litter biomass was the environmental variable that explained most of the variance in species richness. Contrary to what was expected, ant species richness increased in both types of orchards with distance to the forest when other variables were considered. This is possibly due to the fact that the region were the study was conducted is highly deforested and has very few remaining forests. We propose that in areas with little forest cover, orange orchards, whether organic or conventional, provide the type of habitat favored by some species. In sites that are close to real forests, ants would prefer to nest in the forest rather than in the orchards.

Final Thought

This study is one of a few that have been conducted to examine biodiversity within the agricultural matrix in the Amazon region. We study systems that are contributing to the expansion of the agricultural frontier in the Amazon, such as cattle pasture, and soybean plantations, as well as systems that are well established within the areas that were deforested decades ago (agroforestry and orange orchards). All these systems are now embedded within an agricultural matrix that can determine the fate of biodiversity in the highly fragmented areas of the Amazon region (Perfecto et al., 2009- Nature's Matrix, see reference list in Chapter I and II). As agriculture and tree plantations keep encroaching the Amazon forest it is important to understand how biodiversity patterns are affected by the various land use systems and what variables help explain those patterns.

Overall this study found that leaf litter biomass and floristic diversity and biomass of the herbaceous layer were different among all land use systems and were highly correlated with ant species richness, suggesting that they play an important role for the conservation of biodiversity of ants within the agricultural matrices found in the Amazon region.

This study also found that agroforestry systems, orange orchard under organic and conventional management and native forest tree plantations (*paricá*) were the best land use systems (of those examined) for the conservation of ant biodiversity in region. While soybean plantations, cattle pastures, and exotic forest tree plantations (teak), were found to be the worst for ant biodiversity (Figure C.1).

From this study it is also evident that those systems that support the higher ant diversity are those that include trees (agroforestry systems, orange orchards and *paricá* plantations), although not all tree-base systems were beneficial for biodiversity, as demonstrated by the extremely low ant diversity found in teak plantations. Nonetheless, the presence of trees seems to contribute to biodiversity. Therefore, when making decisions regarding land use systems in the Amazon, it is important to consider systems that incorporate trees, such as diverse agroforestry systems. Furthermore, more research should be conducted on diversifying these systems to provide habitat for biodiversity, promote migration between forest fragments, ensure other ecosystem services, such as carbon sequestration and pollination services, and at the same time contribute to improve the livelihood of the people of the Amazon region.