Camila Righetto Cassano

Cobertura florestal e intensificação do manejo: desafios para manutenção de biodiversidade em mosaicos agroflorestais

Forest cover and management intensification: challenges for biodiversity maintenance in agroforestry mosaics

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Tese apresentada ao Instituto de Biociências da Universidade de São Paulo, para obtenção de Título de Doutor em Ciências, na área de Ecologia.

Orientadora: Renata Pardini

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Prof(a). Dr(a). Renata Pardini
Orientadora

"Noite de escuro não serve

Pra caçar de madrugada

Caçador dá muito tiro, o iaiá

De manhã não acha nada

Veado corre é pulando

Cotia corre na trilha

Se eu fosse governador

Ah, Meu Deus, manobrasse a Bahia

Isso que tu tá fazendo

Comigo tu não fazia, camará

Iê, é hora é hora..."

(ladainha de capoeira, Mestre Paulo dos Anjos)

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Às florestas do sul da Bahia que, com sua beleza e espíritos, me inspiraram a trabalhar pela conservação.

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RESUMO

Sistemas agroflorestais abrigam grande biodiversidade quando comparados a outros usos da terra e representam uma ferramenta importante para conciliar produção agrícola e conservação de recursos naturais em paisagens modificadas pelo homem. Esta tese é composta de três capítulos que sintetizam e ampliam o conhecimento a respeito da influência do manejo das agroflorestas e da cobertura florestal remanescente sobre a distribuição dos organismos e apontam ações visando a conservação de biodiversidade em mosaicos agroflorestais. No primeiro capítulo, revisamos a informação científica publicada até 2008 sobre a diversidade da fauna e flora nativas nas agroflorestas de cacau do sul da Bahia. Com base nesta síntese, identificamos recursos chave presentes nessas plantações que se relacionam com a maior riqueza de espécies ou presença de táxons ameaçados e evidenciamos a importância da manutenção de florestas nativas em mosaicos agroflorestais. Finalizamos esta revisão com recomendações para um manejo adaptativo visando a conservação de biodiversidade em propriedades e mosaicos agroflorestais e levantamos questões a serem respondidas por estudos futuros. Nos capítulos 2 e 3, apresentamos dois estudos empíricos realizados a partir da amostragem de mamíferos por armadilhas fotográficas em 30 agroflorestas e remanescentes florestais adjacentes a nove delas. No segundo capítulo, construímos e testamos hipóteses que descrevem quais espécies de mamífero de maior porte com ocorrência na região devem ser positiva e negativamente influenciadas pela conversão de florestas em agroflorestas e como estas respostas diferenciais deveriam se refletir na estrutura, composição e diversidade das assembléias. O valor de conservação das agroflorestas é apoiado pela presença de espécies endêmicas ou ameaçadas de extinção e pela composição similar de espécies entre florestas e agroflorestas. Entretanto, ao identificar que espécies caçadas e mortas por retaliação são raras ou ausentes na nossa amostragem e que agroflorestas são menos usadas por espécies arborícolas e mais usadas por espécies exóticas do que as florestas, nós explicitamos porque o valor de conservação das agroflorestas deve ser visto com cuidado, destacando questões que podem contribuir para a viabilidade das populações de mamíferos de maior porte em paisagens agroflorestais. No terceiro capítulo, contrastamos a importância relativa da intensificação do manejo e da cobertura florestal do entorno sobre o uso das agroflorestas de cacau por mamíferos de maior porte. Nossos resultados indicam que, na paisagem de estudo, fatores diretamente e indiretamente relacionados à intensificação do

manejo das agroflorestas são mais importantes para a distribuição das espécies de mamíferos do que a quantidade de florestas remanescentes no entorno. A taxa de registro de cães domésticos (fator indireto) teve efeito negativo sobre várias espécies de mamíferos, enquanto a redução da conectividade do dossel das agroflorestas (diretamente associada ao manejo do sombreamento) teve um efeito negativo sobre espécies arborícolas e a redução da altura do estrato herbáceo (diretamente associada à freqüência de roçagem) um efeito positivo para uma espécie generalista. Por outro lado, nesse mosaico agroflorestal com grande quantidade de florestas nativas remanescentes, apenas as espécies generalistas responderam à variação da quantidade de florestas no entorno, sendo mais comuns em agroflorestas circundadas por menor quantidade de floresta. Assim, em mosaicos agroflorestais com elevada cobertura florestal, restringir os efeitos negativos decorrentes da intensificação do uso da terra, como o controle das populações de cães domésticos e das atividades de caça e o planejamento cuidadoso do sombreamento das agroflorestas, deve resultar em maiores benefícios para o uso das agroflorestas por mamíferos de maior porte do que alterações na configuração espacial dos remanescentes florestais. Entretanto, a importância da cobertura florestal em relação à intensificação do manejo deve aumentar em paisagens mais desmatadas. Em conjunto, os três capítulos desta tese mostram que o conhecimento científico atual pode embasar o desenho de práticas e políticas públicas para conciliar produção agrícola e conservação em mosaicos agroflorestais. Tais práticas devem ser continuamente avaliadas, contribuindo e sendo repensadas com o avanço do conhecimento científico.

ABSTRACT

Agroforestry systems harbor high levels of biodiversity if contrasted to other land uses and represent an important tool to reconcile agricultural production and conservation in human-modified landscapes. This thesis is composed of three chapters that synthesize and expand the knowledge on the influence of agroforest management and remaining forest cover on the distribution of organisms, and highlight practices aiming biodiversity conservation in agroflorestry mosaics. In the first chapter we review the scientific information published until 2008 on the diversity of native fauna and flora in cacao agroforests in southern Bahia. Based on this synthesis, we identify key agroforest resources associated with higher species richness or the presence of threatened taxa, and emphasize the importance of forest cover maintenance in agroforestry mosaics. We end this review with recommendations to an adaptive management aiming biodiversity conservation on farms and agroforestry mosaics and consider issues to be developed by future studies. On chapters 2 and 3 we present two empirical studies based on photographic records of large mammals in 30 agroforests and forest remnants adjacent to nine of them. In the second chapter we constructed and tested hypotheses describing which mammals from the regional species pool should be positively and negatively influenced by the conversion of forest to agroforests and how these differential responses should change assemblage structure, composition, and diversity. The high conservation value of agroforests is supported by the presence of species of conservation concern, and a similar species composition between forests and agroforests. However, by identifying that hunted and persecuted species are rare or absent in our samples, and that agroforests are less used by arboreal species and more used by non-native species than forest, we explicit why the conservation value of agroforests should be viewed with caution and underline issues that should be addressed to improve the viability of large mammal populations in agroforestry mosaics. In the third chapter we contrast the relative importance of management intensification and surrounding forest cover for the use of agroforests by large mammals. Our results indicate that, in the studied landscape, factors directly and indirectly related to management intensification are more important to define large mammal distribution than the amount and spatial arrangement of surrounding forest cover. High domestic dog capture rate (an indirect factor) had a negative effect on several species, while low canopy connectivity (directly associated to shade management) was negative correlated to the distribution of arboreal

species and lower herbaceous vegetation (directly associated to ground vegetation clearance) had a minor positive influence on generalist species. On the other hand, in this agroforestry mosaic highly covered by native forests, only generalist species responded (positively) to the decrease in surrounding forest cover. In agroforestry mosaics from highly forested regions, restricting the negative effects of land-use intensification, such as the control of domestic dog populations and overhunting, and the careful planning of shade management, is likely to bring greater benefit to large mammal use of agroforests than the spatial arrangement of forest remnants. However, the importance of forest cover relative to management intensification is likely to be higher as deforestation increases. The three chapters show that current scientific knowledge can already support policy and management practices to conciliate agricultural production and biodiversity conservation in agroforestry mosaics. Such practices should be continuously evaluated, contributing to and being supported by improvement of scientific knowledge.

INTRODUÇÃO GERAL

A conciliação entre o aumento da produção agro-pecuária, que depende do avanço da conversão de vegetação nativa ou da intensificação do manejo dos agroecossistemas, e a conservação de biodiversidade e serviços ambientais representa um dos grandes desafios para a humanidade neste século (Sala *et al.* 2000, Foley *et al.* 2005, MEA 2005). A conversão de vegetação nativa e a intensificação do uso do solo têm impactos especialmente relevantes sobre as florestas tropicais, onde se concentra a maior parcela das espécies terrestres (Sala *et al.* 2000, Laurance 2007). A demarcação de áreas protegidas por lei, sejam elas Unidades de Conservação, Reservas Legais ou Áreas de Preservação Permanente, é uma ferramenta importante para a manutenção da biodiversidade e serviços ambientais que contribuem para o bem estar humano. Entretanto, esses ecossistemas protegidos são influenciados por perturbações das áreas adjacentes e sua manutenção em longo prazo depende de processos ecológicos que acontecem em escalas espaciais mais amplas (Chazdon *et al.* 2009, Gardner *et al.* 2009). A necessidade de maximizar os ganhos ambientais em paisagens dominadas por ecossistemas modificados pelo homem tem impulsionado o desenvolvimento de disciplinas como ecologia de paisagens, biologia da conservação e ecologia da restauração (Ehrlich 2007). Sínteses do conhecimento em tais áreas têm possibilitado a construção de princípios gerais para nortear o manejo de paisagens (Lindenmayer & Hobbs 2007, Lindenmayer *et al.* 2008, Gardner *et al.* 2009). No entanto, a compreensão de peculiaridades regionais, tanto no que diz respeito aos ecossistemas naturais quanto ao histórico de ocupação e cenário sócio-político-econômico atual, são fundamentais para que tais princípios sejam implementados com sucesso (Fischer *et al.* 2008, Gardner *et al.* 2009, Gardner *et al.* 2010).

A redução da quantidade de vegetação nativa em uma paisagem influencia por si só a biota, pois representa uma diminuição da quantidade de habitat e recursos disponíveis para as espécies restritas a esse(s) ecossistema(s). Além disso, quanto menor a quantidade de vegetação remanescente, maior a tendência de formação de fragmentos de habitat, que se tornam menores e mais distantes entre si à medida que novos desmatamentos acontecem (Fahrig 2003). Fragmentos de vegetação mantêm números reduzidos de espécies se comparados a trechos contínuos, pois: (1) constituem um recorte (uma "amostra") da vegetação contínua que, sobretudo em sistemas tropicais, são heterogêneas e com muitas espécies distribuídas em machas; (2) possuem populações pequenas e, portanto, mais

suscetíveis à extinção local em conseqüência de estocasticidade demográfica e ambiental e do aumento da endogamia; (3) possuem grande extensão de borda exposta a outros ambientes onde as condições ambientais resultam em alterações na estrutura da vegetação, e na composição e dinâmica da biota (Laurance 2008). Se funcionalmente conectados, o conjunto de fragmentos pode manter populações viáveis - em sistemas de metapopulações - e assim um maior número de espécies e comunidades mais semelhantes aos existentes em trechos de vegetação contínua (Hanski 1998, Pardini *et al.* 2010).

Entretanto, a distribuição das espécies e a manutenção de processos ecológicos dependem não apenas da quantidade e disposição espacial da vegetação remanescente, mas também dos usos da terra que, em conjunto, constituem a matriz de uma paisagem fragmentada (Laurance 2008). Quanto mais similar ao habitat nativo, maior será a probabilidade dos ambientes da matriz: (1) constituírem habitat para espécies nativas; (2) aumentarem a conectividade funcional entre fragmentos de vegetação; e (3) reduzirem a diferença do microclima nas bordas em relação ao interior do fragmento. Os diferentes usos da terra influenciam também a intensidade e distribuição de distúrbios (e.g. ocorrência e expansão de incêndios), a distribuição de espécies invasoras e a pressão humana sobre os sistemas naturais na paisagem (Kupfer *et al.* 2006). A relevância da matriz de ambientes alterados sobre a resiliência de paisagens modificadas pelo homem tem recebido cada vez mais atenção de cientistas e atores envolvidos no planejamento e manejo do uso da terra (Lindenmayer & Hobbs 2007) e despertado o interesse sobre usos da terra menos intensivos, que mantém maior biodiversidade associada (Tscharntke *et al.* 2005).

A intensificação do uso da terra está no centro de uma discussão sobre o planejamento de paisagens visando a conciliação entre produção agrícola e conservação de biodiversidade e serviços ecossistêmicos. Por um lado, sistemas de manejo extensivo (*biodiversity-* ou *wildlife-friendly*) podem beneficiar a conservação da biodiversidade ao constituir habitat e promover conectividade entre remanescentes de habitat (Perfecto & Vandermeer 2008). Por outro lado, sistemas de manejo mais intensivo podem atingir a mesma produção líquida em extensões territoriais menores, e com isso permitem que maiores extensões de terras sejam mantidas com vegetação nativa (Balmford *et al.* 2005). Ainda que parte dessa questão esteja ligada a cenários sócio-político-econômicos que se adéquam ou conduzem a produções mais ou menos intensivas (Fischer *et al.* 2008, Vandermeer 2011), a adoção de uma ou outra estratégia deve considerar também características biofísicas das paisagens

(ex: topografia, qualidade do solo, heterogeneidade espacial dos ecossistemas) e a resposta das espécies às modificações ambientais (Fischer *et al.* 2008).

Sistemas agroflorestais são usos da terra onde espécies lenhosas estão presentes em uma mesma unidade de manejo com cultivos agrícolas e/ou animais (Somarriba 1992). A manutenção de árvores faz com que esses sistemas exibam condições mais similares às florestas nativas do que outros usos da terra, oferecendo assim possibilidades para conciliar produção e conservação em paisagens florestais fragmentadas (Schroth *et al.* 2004, Schroth & Harvey 2007, Perfecto & Vandermeer 2008). Em regiões tropicais, grandes extensões de florestas nativas foram convertidas em agroflorestas de cacau (*Theobroma cacao*) e café (*Coffea* spp.), cultivos que se desenvolvem preferencialmente na sombra (Rice & Greenberg 2000, Bhagwat *et al.* 2008). Essas regiões com grandes extensões de agroflorestas oferecem uma oportunidade para a realização de pesquisas que visam avaliar a influência de usos da terra extensivos sobre a diversidade de espécies e processos ecológicos.

Na região sul da Bahia, aproximadamente 6000 km^2 de terras já foram ocupados por sistemas agroflorestais destinados à produção de cacau com sombreamento proporcionado por árvores nativas, regionalmente conhecidos como *cabrucas* (dado para os anos 1996/97; Landau *et al.* 2008). Parte dessas *cabrucas* tem sido gradualmente substituída por outros usos da terra ou sistemas de produção mais simplificados, confirmando uma tendência prevista no final da década de 90 (Johns 1999), quando a queda da produtividade das lavouras cacaueiras e do valor do cacau no mercado internacional resultou no abandono de muitas fazendas. No entanto, trechos dessa região que ainda abrigam áreas representativas de *cabrucas* têm sido alvo de estudos importantes visando o conhecimento da diversidade de espécies e viabilidade de populações nesses agroecossistemas e mosaicos agroflorestais (ex: Pardini *et al.* 2009, Cassano *et al.* 2011, Oliveira *et al.* 2011, Capítulo 1). A grande diversidade de espécies e alto grau de endemismos da floresta Atlântica nesta região, a pressão por parte de movimentos sociais para ocupação de propriedades rurais abandonadas e a necessidade de recuperação da lavoura cacaueira fizeram com que o sul da Bahia fosse também uma área de interesse para o desenvolvimento de projetos de conservação e desenvolvimento agrário por parte de diversas organizações não governamentais.

O presente estudo é fruto de um projeto iniciado em 2006 pelo Instituto de Estudos Socioambientais do Sul da Bahia (IESB) em parceria com a Birdlife International e a

Sociedade para Conservação das Aves do Brasil (SAVE-Brasil). Esse projeto teve o objetivo de promover práticas agrícolas sustentáveis em propriedades rurais localizadas no Corredor Una-Lontras, região que compreende as terras no entorno e entre duas Unidades de Conservação da região sul da Bahia: a Reserva Biológica de Una e o Parque Nacional das Serras das Lontras. Dentre as práticas incentivadas estavam a recuperação de sistemas agroflorestais destinados à produção de cacau, o fomento à produção orgânica, a averbação de Reserva Legal e o incentivo à criação de Reservas Particulares do Patrimônio Natural. Dado o interesse em incentivar ações que comprovadamente contribuíssem para a conservação da biodiversidade regional, o projeto promoveu também o início de uma compilação de dados científicos e o desenvolvimento de pesquisas que investigassem relações entre o mosaico agroflorestal construído e mantido pelas práticas incentivadas e a diversidade de grupos biológicos, entre eles os mamíferos.

A compilação de dados científicos foi ampliada ao longo do período em que elaborei o plano de pesquisa para o exame de ingresso no programa de pós-graduação em Ecologia, e ao longo do primeiro ano do curso. Com o apoio de pesquisadores da ONG Conservação Internacional (CI-Brasil) e da Universidade Estadual de Santa Cruz (UESC) esta revisão de literatura se transformou num artigo científico, que corresponde ao primeiro capítulo da presente tese de doutoramento. A revisão de literatura foi dividida em três partes. Na primeira, revisamos estudos que comparam a diversidade de espécies das agroflorestas de cacau com outros tipos de vegetação e relacionam as diferenças encontradas com o contexto da paisagem onde esses sistemas se encontram. Em seguida, tratamos dos resultados de estudos que relacionam características locais das plantações (influenciadas pelo manejo) com a disponibilidade de recursos para alguma espécie ou grupo biológico. Por último, focamos nos estudos sobre espécies ameaçadas de extinção e na contribuição das *cabrucas* para sua conservação. Na discussão, tecemos recomendações para o manejo das plantações de cacau e para os mosaicos agroflorestais e concluímos com a indicação de gargalos de informação e prioridades para estudos futuros.

O estudo dos mamíferos no mosaico agroflorestal teve seu delineamento aperfeiçoado também durante o período em que elaborei o plano de pesquisa para exame de ingresso no programa de pós-graduação em Ecologia e originou os demais capítulos da tese (2 e 3). O embasamento teórico e a argumentação de ideias desses capítulos receberam contribuições do Dr. Jos Barlow, que se tornou coautor desses capítulos, além da Dr. Renata Pardini, orientadora desta tese. No capítulo 2 construímos hipóteses de como diferentes grupos de

mamíferos de maior porte devem responder às alterações da estrutura da vegetação, grau de perturbação e disponibilidade de alimento que ocorrem quando florestas são transformadas em *cabrucas*, e quais as diferenças esperadas para a estrutura, composição e diversidade da assembléia de espécies nos dois sistemas. Testamos nossas hipóteses usando registros fotográficos coletados em nove *cabrucas* adjacentes a nove remanescentes florestais e partindo de uma lista de espécies conhecidas para a região, que foram previamente classificadas em função do seu hábito alimentar e locomotor e sua vulnerabilidade à caça. No terceiro capítulo, investigamos a importância relativa da cobertura florestal e da intensificação do manejo das *cabrucas* sobre o uso dessas agroflorestas por mamíferos de maior porte. A distribuição da cobertura florestal no entorno dos sítios de amostragem foi caracterizada por variáveis em duas escalas distintas; o manejo foi descrito por duas medidas da estrutura da vegetação e uma medida da intensidade com que cães domésticos visitam as plantações. Utilizamos registros fotográficos dos mamíferos em 30 *cabrucas* e uma abordagem de seleção de modelos para contrastar modelos matemáticos que descrevem diferentes relações de importância das variáveis explanatórias. Com o objetivo de testar se falhas na detecção das espécies interferem nos resultados do estudo, comparamos seleções de modelos que consideram a probabilidade de detecção heterogênea e menor do que um contra modelos que não incluem esse parâmetro.

Os três capítulos centrais da tese são apresentados em formato de artigo científico redigidos em inglês. O primeiro artigo foi publicado em 2009 na revista Biodiversity and Conservation e o segundo submetido para a revista Biotropica em 21 de julho de 2011. A tese é composta ainda por uma discussão geral onde discorro brevemente sobre o avanço do conhecimento sobre biodiversidade em agroflorestas, situando a revisão e os estudos empíricos apresentados nessa tese, retomo os principais resultados e conclusões dos três capítulos e exponho questões que podem contribuir para o avanço do conhecimento e aplicação das agroflorestas como ferramenta para conservação.

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Landscape and farm scale management to enhance biodiversity conservation in the cocoa producing region of southern Bahia, Brazil

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Abstract

In southern Bahia, Brazil, large land areas are used for the production of cocoa (*Theobroma cacao*), which is predominantly grown under the shade of native trees in an agroforestry system locally known as *cabruca*. As a dominant forest-like landscape element of the cocoa region, the *cabrucas* play an important role in the conservation of the region's biodiversity. The purpose of this review is to provide the scientific basis for an action plan to reconcile cocoa production and biodiversity conservation in southern Bahia. The available research collectively highlights the diversity of responses of different species and biological groups to both the habitat quality of the *cabrucas* themselves and to the general characteristics of the landscape, such as the relative extent and spatial configuration of different vegetation types within the landscape mosaic. We identify factors that influence directly or indirectly the occurrence of native species in the *cabrucas* and the wider landscape of the cocoa region and develop recommendations for their conservation management. We show that the current scientific knowledge already provides a good basis for a biodiversity friendly management of the cocoa region of southern Bahia, although more work is needed to refine some management recommendations, especially on shade canopy composition and density, and verify their economic viability. The implementation of our recommendations should be accompanied by appropriate biological and socioeconomic monitoring and the findings should inform a broad program of adaptive management of the *cabrucas* and the wider cocoa landscape.

Keywords: Atlantic forest; biodiversity; *cabruca*; cocoa agroforest; connectivity; fragmentation; landscape management; *Theobroma cacao*

1.1. Introduction

In the face of high rates of tropical forest loss, agroforestry based development models have been proposed as a strategy to conserve natural resources in the tropics (Izac & Sanchez 2001, Schroth *et al.* 2004). The cocoa tree (*Theobroma cacao* L., Malvaceae) is traditionally planted in several tropical regions under a native tree canopy after thinning the original forest (Rice & Greenberg 2000, Ruf & Schroth 2004). Cocoa plantations shaded by native trees are reportedly among the agroforestry practices with greatest potential to reconcile agricultural development and biodiversity conservation (Rice & Greenberg 2000, Schroth *et al.* 2004, Schroth & Harvey 2007). This is particularly relevant in southern Bahia, Brazil's main cocoa production region. Together with the northern part of the state of Espírito Santo, this region forms a center of species endemism within the Atlantic forest biome, where co-generic species from the Amazon forest and from the southern Atlantic forest are found (Brown Jr. 1982, Thomas *et al.* 1998). The region is exceptional for its biodiversity, and in a recent comparison of tree diversity in 22 tropical forests around the world, a forest remnant in southern Bahia was the second richest site (Martini *et al.* 2007). Tree endemism can exceed 25% (Thomas *et al.* 1998). The region also harbors several endemic species of mammals, birds (Bencke *et al.* 2006) and ants (Delabie *et al.* 1998, Lacau *et al.* 2004). However, a recent study estimated that only 6.5% of old growth forest remains in southern Bahia, with few remnants larger than 1000 hectares (Landau 2003).

The cocoa region of southern Bahia is defined here as the area between the Contas and Jequitinhonha Rivers (Figure 1). Cocoa cultivation began here in the eighteenth century and peaked in the 1960s and 1970s. Over the following decades, the government's Executive Commission of the Cocoa Production Plan (CEPLAC) recommended establishing cocoa plantations by felling all native trees followed by planting cocoa seedlings together with bananas and erythrinas (*Erythrina fusca* – an exotic legume with no economic value) as shade trees. However, the lower cost of establishing cocoa groves in the traditional system – partial thinning of the forest and its under-planting with cocoa trees – resulted in the formation of extensive agroforests, known as *cabrucas* (Alger & Caldas 1994, Johns 1999; Figure 2). According to Sambuichi (2006), native tree density in *cabrucas* ranges from 35 to 173 trees per hectare and reaches 355 trees per hectare in abandoned plantations with their vigorous regeneration especially of pioneer species.

In the late 1980s, cocoa cultivation in Bahia entered into a period of decline triggered by falling international cocoa prices and the spread from the Amazon into Bahia of the fungus *Moniliophthora perniciosa*, causal agent of the "witches' broom" disease. The loss of cocoa income led many growers to sell off their shade trees for timber (Alger & Caldas 1994, Araújo *et al.* 2007), while others abandoned their plantations. However, cocoa farms still occupy approximately 6000 km² in southern Bahia (Landau 2003), many of which are abandoned or very extensively managed. Current recommendations by CEPLAC on techniques to rehabilitate cocoa plantations include the grafting of witches' broom resistant cocoa varieties on diseased trees and the replacement of the erythrina shade trees with rubber trees (*Hevea brasiliensis*) in order to increase and diversify farm income (Marques & Monteiro 2006).

Research in southern Bahia and other cocoa producing regions, especially Central America and parts of Africa, has shown that a significant part of the extant native flora and fauna can be conserved in traditional cocoa agroforestry systems (e.g. Rice & Greenberg 2000, Delabie *et al.* 2007, Faria *et al.* 2007, Schroth & Harvey 2007). For example, traditional cocoa agroforests with diversified native shade trees were shown to harbor a richer bird, bat and dung beetle fauna than plantain monocultures in Costa Rica (Harvey *et al.* 2006, Harvey & Villalobos 2007) and a richer ant fauna than pasture in Bahia (Delabie *et al.* 1999). However, these studies have also demonstrated that the contribution of cocoa agro-ecosystems to the conservation of biodiversity is dependent on their structure, composition and management, as well as on the quantity, quality and location of remnants of native forest habitat in the landscape (Schroth & Harvey 2007). Furthermore, there is considerable variation between species groups of conservation concern in Bahia with regard to their ability to utilize shaded cocoa plantations as habitat and to persist in the wider *cabruca* landscape (e.g. Alves 1990, Pardini 2004, Faria *et al.* 2007). Understanding how different elements of the local biota are influenced by site and landscape features is a necessary step for the design of conservation actions.

This article reviews the available information about the biodiversity of cocoa plantations and their landscape context in southern Bahia and identifies recommendations for management actions to improve its conservation. We first review studies that compare the biodiversity of shaded cocoa plantations with that of other vegetation types in the landscape and relate it to the composition of the landscape as a whole. Subsequently, we discuss the specific management of shaded cocoa plantations in relation to patterns of

biodiversity at the site scale. We then focus specifically on endangered species of the region and highlight the contribution that *cabrucas* can make to their conservation. In the discussion we derive recommendations for the conservation management both of cocoa plantations and the wider landscape in southern Bahia, many of which are also applicable in other cocoa regions. We conclude with some research needs.

1.2. Methods

Our information sources were publications in scientific journals as well as reports, Master and Doctoral theses from several Brazilian universities. We also review studies presented at the "First Symposium on the Cocoa Production Landscape and Biodiversity of Southern Bahia" that took place at the State University of Santa Cruz, Ilhéus, Bahia, on October 8 and 9, 2007. A significant part of the body of research we present here is not available in the peer reviewed scientific literature. From a total of 52 scientific studies, we extracted information on the following biological groups: vertebrate fauna (mammals, birds, reptiles and amphibians); invertebrate fauna (butterflies, ants and parasitoids); and flora (trees, ferns and bromeliads). The geographic range of the studies varies from one or a few sites to the entire Bahian cocoa region. Most research was carried out in the counties of Una, Ilhéus and Itabuna, due in part to the location of the main cocoa research institutions there (CEPLAC, University of Santa Cruz) and in part to the importance of cocoa production for these counties. The paucity of information on most of the biological groups as well as differences between studies in survey methods prevented a formal meta-analysis.

1.3. Results

The biodiversity of shade cocoa plantations within the landscape

A main focus of biodiversity research in the cocoa region has been the county of Una surrounding the Una Biological Reserve, which was established in 1980 and is now one of the largest remaining forest blocks in the region (currently 11,000 hectares). The area is still dominated by mature forest remnants (50% of the landscape) interspersed with secondary forests (15%) and *cabrucas* (5%), the remainder being mostly open pastures, crop fields and small plantations (Figure 1). Between 1998 and 2002, researchers of the RestaUna project (http://www.restauna.org.br/) coordinated by the State University of Santa Cruz (UESC), Ilhéus, conducted an inventory of the fauna and flora of the edges and

interior of large (>1000 ha) and small forest fragments (< 200 ha), as well as the *cabrucas* and secondary forests which dominate the surrounding matrix. For the purpose of the study, the latter habitat was defined as at least 15 year old forest regrowth after complete clearing of a site. This research revealed the high diversity of assemblages of all investigated biological groups (small mammals, bats, birds, leaf-litter frogs and reptiles, frugivorous butterflies, ferns and bromeliads) in the *cabrucas* and secondary forests, with species richness and abundance values comparable to those in the interior of mature forest fragments. There were, however, changes in the species composition and relative abundance of different species within each group, indicating an idiosyncratic response of each biological group to the alterations in the landscape that cocoa growing and other land-uses had brought about (Table 1). Here we summarize the salient results for each taxonomic group.

Small mammals: For small mammals, the abundance of terrestrial and scansorial species that dominate the communities of the forest interior (*Oryzomys laticeps*, *Marmosops incanus* and *Monodelphis americana*) showed a relative decrease, while the abundance of three arboreal species (*Marmosa murina, Rhipidomys mastacalis* and *Micoureus demerarae*) increased in the disturbed habitats, i.e. forest edges, *cabrucas* and secondary forests. Two species that are typical of open environments, *Oligoryzomys* sp. and *Akodon cursor*, were found in the three disturbed habitats but were rare or absent from the forest interior (Pardini 2004). Possibly the increased richness of terrestrial small mammals in the disturbed habitats, including the *cabrucas*, narrowed the niche for the corresponding forest interior species through competition, but did not affect arboreal species (Pardini 2001).

Bats: Faria (2006), Faria *et al.* (2006), Faria & Baumgarten (2007) showed that the richness and abundance of bats were higher in *cabrucas* than in natural forest. This was even true for gleaning insectivores, a feeding guild usually regarded as sensitive to habitat fragmentation (Fenton *et al.* 1992) and disturbance (Medellin *et al.* 2000). *Rhinophylla pumilio*, a dominant bat species in large forest remnants that is considered vulnerable to fragmentation (Henry & Kalko 2007) was the only species whose capture frequency was lower in *cabrucas*, where *Carollia perspicillata,* the second most abundant species in the large forest remnants, was the dominant species. This contrasting response of *R. pumilio* and *C. perspicillata* to *cabruca* versus forest was confirmed in the neighboring county of Ilhéus that is much poorer in forest and richer in *cabrucas* than Una (Faria *et al.* 2006). In Una, bat abundance and richness were inversely correlated to the understory density of

forest patches (Faria 2002), suggesting that the simpler vegetation structure in *cabrucas* facilitated their flight (Faria *et al.* 2006, Faria & Baumgarten 2007). This open understory may also explain why bat species that normally use the forest canopy, such as *Artibeus jamaicencis*, *A. lituratus*, *Chiroderma villosum* and *Phyllostomus hastatus*, were caught in mist nets at 1-2 m above the ground in *cabrucas*. In another study in neighboring Ilhéus, *Artibeus obscurus*, a forest-dwelling species negatively affected by decreasing fragment size in the Una region (Faria 2006), was more abundant in cocoa plantations than in forest irrespective of their distance to forest and canopy cover, presumably due to the greater density of fruit trees such as *Ficus* spp. on which they feed (Farias & Faria 2007).

Avifauna: In forest-rich Una, six typical forest bird species - the understory insectivores *Drymophila squamata*, *Herpsilochmus pileatus*, *Pyriglena leucoptera*, *Rhytipterna simplex* and the terrestrial insectivores *Conopophaga melanops* and *Formicarius colma* - were less abundant in *cabrucas* than in forest and one forest bird (*Lipaugus vociferans*, an understory frugivore) was absent from *cabrucas* (Faria *et al.* 2006). For birds, *cabrucas* seem to be transitional habitat where species typical of forest, fragment edges and open areas occur together, with the latter probably benefiting from the simpler vegetation structure (Faria *et al.* 2006).

Leaf-litter herpetofauna: Dixo (2001) found no differences in total species richness or abundance of litter frogs and lizards between *cabrucas* and forest fragments between 25 and >1000 ha in size in Una, but reported significant differences in the abundance of two lizard species, with greater abundance of *Leposoma scinconoides* and lower abundance of *Enyalus catenaci pictus* in *cabrucas* than forest. The latter species was also less abundant in secondary than in primary forest. (It should be mentioned that many species found in both habitats were rare and it is generally difficult to find significant differences for such species.)

Nymphalid butterflies: The *cabrucas* of Una harbored fewer species of the Morphinae sub-family than forest fragments between 25 and >1000 ha in size and the abundance of two species from the Brassolinae sub-family, *Caligo idomenaeus* and *Eryphanis polyxena*, tended to be lower in *cabrucas* than in these forest fragments. Large forest species from the Euritelinae, Coloburinae and Satyrinae sub-families showed no difference in species richness between *cabrucas* and forest. *Cabrucas* seemed to be inhospitable to some of the species that are considered invaders of disturbed areas in the region (such as *Biblis hyperia*

and *Caligo illioneus*), and only *Yphtimoides reneta* and *Hermeuptychia hermes* from this group were found in significant numbers (Accacio 2004).

Ferns and bromeliads: The fern communities of *cabrucas* in Una were poorer in numbers of species than forest interiors and some species (including those of the Hymenophyllaceae family) were missing. On the other hand, *cabrucas* were suitable for other typical forest ferns, such as *Adiantum diogoanum,* as well as fern species typical of disturbed habitat (Paciencia & Prado 2004, 2005a, 2005b). The discovery of a new species of fern, *Adiantum discolor*, in a *cabruca* during this study (Prado 2000) highlights the importance of these agroforests for biodiversity conservation. Epiphytic bromeliads were significantly less abundant and less species rich in *cabrucas* than in forest fragments (both interior and edge; Alves 2005). Understory bromeliads from the forest interiors were found to be heavily affected by the conversion of forest to *cabrucas*: from nine species reported in the interiors of forest fragments (*Aechmea mollis, A. turbinocalyx, Areococcus paviflorus, Lymania globosa, L. azurea, L. south bahiai, Nidularium amorimii, Vriesea drepanocarpa* and *V. duvaliana*) none was found in *cabrucas*. All except the latter two of these species are endemic to the Atlantic Forest of southern Bahia and northern Espírito Santo. Three bromeliad species from the forest canopy (*Aechmea conifera, A. leonard-kentiana* and *Aechmea* sp.) were either absent or very rare in *cabrucas*, while another four canopy species (*Aechmea lingulata, Honhenbergia brachycephala, Vriesea* sp. and *V. procera*) were found exclusively in *cabrucas*. The most abundant bromeliad species in *cabrucas* was *Aechmea lingulata* whose large range stretches from Central America to southeast Brazil (Alves 2005).

The research reviewed thus far, most of which was carried out under the umbrella of the RestaUna project, thus showed that the *cabrucas* of forest-rich Una are highly species diverse, harboring about 70% of the 431 species of ferns, frugivorous butterflies, litter frogs and lizards, small mammals, birds and bats that were found in all vegetation types (including mature forest) together in this project.

In subsequent work, the same biological groups except for bromeliads were surveyed in primary forest remnants and *cabrucas* about 30-40 km further north in the county of Ilhéus (Figure 1). Although the forests of Una and Ilhéus originally belonged to a single block before the agricultural transformation of the region, the two landscapes are now markedly different. While *cabrucas* occupy 5% and forest 50% of the Una landscape, the landscape of Ilhéus is dominated by *cabrucas* (82%) with only 5% under forest.

Overall, the biological communities were poorer in the *cabruca*-dominated landscape of Ilhéus than in the forest-dominated landscape of Una, with fewer species of small mammals (Ortiz *et al.* 2002), bats (Faria *et al.* 2006, Faria & Baumgarten 2007), birds (Faria *et al.* 2006), ferns, and litter herpetofauna (Faria *et al.* 2007; Figure 3). This suggests that despite the relatively high habitat quality of the *cabrucas* that dominate the landscape, the amount of natural habitat remaining in the Ilhéus landscape is too low to retain completely the original species assemblages. Also, a greater variation in bird communities between individual forest fragments was observed in forest-poor Ilhéus than in forest-rich Una, suggesting that the connectivity provided by the *cabrucas* between forest fragments may be insufficient to allow rapid recolonization of fragments following local extinctions (Faria *et al.* 2007).

The important contribution of the *cabrucas* and the vital role of forest remnants for maintaining the region's biodiversity also emerge from other research. In a study by Alves (1990), the species richness of medium and large sized birds and mammals was greater in two *cabrucas* close to large forest remnants than in *cabrucas* that were distant from forest fragments, suggesting that the forest remnants served as a refuge for these animals. In this study, typical understory birds of the Formicaridae and Pipridae families were less common in *cabrucas* than in forest while ground-dwelling species of the Cracidae and Tinamidae families where almost absent, confirming that *cabrucas* offer habitat for some but not all forest birds. Among mammals, the collared peccary (*Pecari tajacu*), deer (*Mazama* sp.) and two large primates (the yellow-breasted capuchin monkey, *Cebus xanthosternos*, and the southern Bahian masked titi monkey, *Callicebus melanochir*) were rarely recorded in *cabrucas*, while small primates (Wied's black-tufted-ear marmoset, *Callithrix kuhlii*, and the golden-headed lion-tamarin, *Leontopithecus chrysomelas*) as well as generalist species such as tayra (*Eira barbara*) and crab-eating raccoon (*Procyon cancrivorus*) were more common. Alves (1990) reported that the occurrence of *L. chrysomelas* in *cabrucas* was positively related to the proximity of forest remnants and their size. These observations are supported by a camera-trap study where golden-headed lion-tamarins were recorded in five out of 20 *cabrucas* and in five out of seven forest fragments, with all *cabrucas* that harbored tamarins being close to forest remnants (Cassano 2007). On the other hand, groups of lion tamarins that live continuously in *cabrucas* without access to forest are also known (Leonardo Oliveira, pers. com. 2008). In another study comparing *cabrucas*, secondary forests and young fallows throughout the

region, Neves *et al.* (2007) recorded *C. kuhlii* and *L. chrysomelas* in all three habitats (the two species were found in, respectively, 72% and 32% of the *cabrucas* visited), but did not find large-sized primate species such as howler monkey (*Alouatta guariba*) and yellow-breasted capuchin monkey in *cabrucas*.

Deforestation and the increase in cultivated areas in the southern Bahian landscape were suggested as causal factors in the replacement of the leafcutter ant, *Atta cephalotes,* a typical species of forest environments, by *Atta sexdens*, a species more typical of degraded areas in this region (Delabie 1990, Delabie *et al.* 1999, Delabie *et al.* 2007). Forest loss is also held responsible by these authors for the replacement of some ant genera previously recorded in *cabrucas*, such as the decrease in abundance of species of the Ponerinae and Ecitoninae sub-families and the concomitant increase in species of the Formicinae and Dolichoderinae. According to Delabie *et al.* (2007), several species of army ants of the genus *Eciton* can be found in *cabrucas*, provided there are contiguous forest remnants to maintain their colonies. Open areas, such as pastures and clearings, are barriers to foraging for some army ant species. A predictable consequence of the decline in army ants (*Eciton* and *Neivamyrmex*) in the cocoa region would be the disappearance of their associated fauna, especially ant birds (Delabie 1990, Delabie *et al.* 2007) and many invertebrates that benefit from the niches created by the periodic elimination of litter fauna along ant trails (Franks 1989).

Biodiversity conservation and cocoa management within the cabruca *system*

The conversion of forest into *cabrucas* implies a major alteration of plant species composition and forest structure, including the clearing of the forest understory which hosts most of the forest plant species (Martini 2007) and the loss of over 90% of the overstory trees (Alves 1990). Nevertheless, botanical surveys in *cabrucas* have found large numbers of native canopy tree species, including timber species such as *Cedrela odorata*, *Nectandra* sp. and *Cariniana* spp. that have become rare in unprotected natural forest due to over-use (Vinha & Silva 1982, Sambuichi 2002, 2006, Lobão 2007). For example, Sambuichi & Haridasan (2007) found 293 morpho-species of trees in five *cabrucas* in Ilhéus county, and Rolim & Chiarello (2004) inventoried 105 species in a *cabruca* in the neighboring state of Espírito Santo. Considering the extremely high tree diversity of southern Bahia (Thomas *et al.* 1998), this is an important conservation service provided by

cabrucas. However, common management practices in *cabrucas* tend to decrease tree diversity over time. These include the progressive thinning of shade canopies – partly motivated by official recommendations to keep only 25-30 trees per hectare to maximize cocoa yields (Alvin 1966, Alvin & Pereira 1970) – and the substitution of old forest trees – typically climax and late secondary species – by faster growing pioneer and economically valuable species, often exotics (Sambuichi 2006, Mota *et al.* 2007, Sambuichi & Haridasan 2007). The reduction of shade tree density and diversity is likely to negatively affect habitat conditions for many fauna species that depend on the microclimate, food and other habitat conditions created by the shade canopy, although this relationship has not been well studied. For example, shade trees in *cabrucas* provide food for fruit and nectar-eating bats (Faria & Baumgarten 2007) and can be used for displacement and food by two endangered arboreal mammals, the golden-headed lion-tamarin (*Leontopithecus chrysomelas*) and the maned three-toed sloth (*Bradypus torquatus*; see following section). Greater diversity of shade trees in cocoa plantations was also positively related to parasitoid richness and abundance and thus supported natural pest control (Sperber *et al.* 2004).

Common management practices of *cabrucas* also impact herbaceous and understory plant species. According to Martini (2007), approximately two thirds of the plant species in forest fragments in southern Bahia are understory species. Except for species that re-sprout after cutting, these species are eliminated by the establishment and periodic weeding of cocoa plantations, and therefore depend on natural forest areas for their conservation. Like canopy trees, understory plants are also an important basis for fauna species in *cabrucas*. Heliconias and a large number of Marantaceae, which resprout between weedings, account for the diversity and abundance of small butterflies of the Satyrinae sub-family in the *cabrucas* of Una (Accacio 2004). The leaves of *Heliconia* spp. in *cabrucas* are also used as shelter by bats (e.g. *Thyroptera tricolor*), while other bat species use termite nests and tree holes (e.g. *Lophostoma sivicolum* nests in active termite hills; Faria & Baumgarten 2007).

According to Faria *et al.* (2007), epiphytic ferns are less abundant in *cabrucas* than in forest, possibly because they are removed from the trees by farm workers. The pruning of cocoa trees and removal of epiphytes from their stems and branches are part of the normal cocoa management and may reduce the diversity of epiphytes in the *cabruca* understory (Alves 2005). Micro-climatic modifications such as lower atmospheric humidity and lack of suitable dispersers probably also reduce the richness of the bromeliad flora in *cabrucas*,

although some species with a high vegetative reproduction capacity seem to do well (Alves 2005).

The suspended soil in the shade canopies of *cabrucas* provides a rooting substrate for bromeliads and other epiphytes (Figure 4) and thus helps to maintain a rich fauna of ant species that are either arboreal or normally live in the litter layer, in addition to many other invertebrate species (Delabie 2003). Cocoa plantations shaded by *Erythrina* sp. are considered good ant habitat because the architecture of these leguminous trees favors the development of a great number of epiphytes (J.H.C. Delabie, per. obs.). The same can be said for tree snakes that use epiphytes as shelter and feeding sites (A.J.S. Argôlo, pers. com. 2007).

The lower abundance or even absence of understory birds and small arboreal mammals in *cabrucas* compared to forest can partly be explained by differences in vegetation structure, especially the simplification of the understory and absence of connecting vines between understory and tree canopies (Alves 1990, Moura 1999, Faria *et al.* 2006). Cassano & Moura (2003) found a possible relationship between vegetation structure and small mammal diversity among cocoa plantations. They recorded five small mammal species in the simplified environment of a cocoa plantation shaded by rubber trees (*Hevea brasiliensis*, Euphorbiaceae) with complete suppression of the herbaceous stratum; seven species in a similar plantation where the herbaceous stratum was maintained; and 12 species in a *cabruca* with complex shade canopy whose understory had not been slashed for three years. The structural complexity of the vegetation was also suggested by Pinto *et al.* (1993) to be a determinant of the richness and abundance of small mammals in a comparison of native forest, an oil palm plantation, a cocoa plantation shaded by rubber trees, and a *cabruca*.

Majer *et al.* (1994) found that arboreal ant communities in the understory of three cocoa plantations shaded by erythrinas showed stronger dominance of species of the genera *Azteca*, *Ectatomma* or *Crematogaster* than the understory of a *cabruca*, despite similar overall ant diversity. They explained the difference with greater diversity of the shade canopy, greater plantation age and no insecticide use in the *cabruca*. The *cabruca* was the only of the four plantations where the "little fire ant" *Wasmannia auropunctata* was not recorded. This species shows great capacity for population growth and colonization of new areas where other ant species were eliminated by insecticides (Delabie 1990). Insecticide use and the removal of nests of social ants in *cabrucas* are held responsible for the

replacement of ants of the genus *Solenopsis* and, in the tree stratum, *Azteca* spp. by *W. auropunctata* (Delabie 1990). In a more recent study, Delabie *et al.* (2007) showed that *W. auropunctata* has a common but discrete presence in forest remnants where it is limited to small populations in the litter stratum, while in cocoa plantations it colonizes the tree canopies to raise sucking insects and excludes other ant species. Delabie *et al.* (2007) emphasized the importance of maintaining an intact litter layer in cocoa plantations because, similar to forest, this stratum concentrates the highest ant diversity. Pruning of the cocoa canopies and the reduction of their height to control infestations by the witches' broom disease are singled out for their negative effect on the arboreal ant communities which consequently cease to play their role in the biological control of herbivorous insects that damage the cocoa trees (Delabie & Mariano 2001, Delabie *et al.* 2007). The same authors suggested that the rarity of the forest ant *Typhlomyrmex rogenhoferi* in cocoa plantations in southern Bahia is a consequence of the rarity of fallen tree trunks where these ants build their nests (Delabie *et al.* 2007).

Contribution of cabrucas *to the conservation of endangered species*

Cabrucas provide habitat and resources to several endangered endemic species of the Atlantic forest and should therefore be considered in action plans for their conservation.

Mammals: At least two endangered mammal species have been recorded in *cabrucas*: the golden-headed lion-tamarin (*Leontopithecus chrysomelas*; Alves 1990, Raboy *et al.* 2004, Cassano 2007, Neves *et al.* 2007) and the maned three-toed sloth (*Bradypus torquatus*; Cassano 2006). Two groups of lion-tamarins that were monitored in the Una Biological Reserve used a nearby *cabruca* in their daily activities and as sleeping sites (Raboy *et al.* 2004). The tamarins preferentially used tree holes as shelter and the greater frequency of suitable holes seemed to be a reason why they slept more often in mature forest than in secondary forest and *cabrucas* (Raboy *et al.* 2004). Oliveira *et al.* (2007) listed 118 tree species used by these same groups of lion tamarins in the Una Biological Reserve between 1998 and 2006, with 80 species being used for food and 63 species as sleeping sites. Such information on preferred tree species could be used for improving the habitat quality of *cabrucas* and other farm areas for this endangered primate (and similarly other fauna species), thereby contributing to their more effective conservation. In addition, bromeliads of the genus *Aechmea*, including *A. conifera* and *A.* cf. *depressa*, were used as food and

foraging sites by the tamarins (Nascimento *et al.* 2007) and were their main food item in abandoned *cabrucas* in the Una Biological Reserve (Catenacci 2008).

Cassano (2006) monitored a female maned three-toed sloth, *Bradypus torquatus,* an endemic species of the Atlantic forest, and its offspring in a *cabruca* near the Una Biological Reserve. The sloths used the *cabruca*, which had an unusually dense canopy, as their main habitat and the most abundant tree species in the shade canopy were part of their diet (Correia *et al.* 2006, Barreto & Cassano 2007).

The cocoa region of southern Bahia is also the main area of occurrence of the endemic painted tree rat, *Callistomys pictus*. The *cabrucas* are probably important habitat for this rodent given that they are the main environment where the species has recently been recorded. Hollow trees and epiphytic bromeliads are apparently used as shelter. Cocoa leaves and fruits are consumed, but the species does not appear to cause significant economic damage (Moura 2005, 2008). However, this rodent is being captured and killed by farm workers, suggesting a need for local education programs to enhance the conservation of the species.

Birds: Faria *et al.* (2006) reported the occurrence of the pileated antwren (*Herpsilochmus pileatus*) in *cabrucas* both in the forest-dominated Una and the *cabruca*-dominated Ilhéus counties. However, in the latter area the species was very rarely found, indicating that despite its ability to utilize *cabrucas* it is vulnerable to forest loss in a landscape-scale. The *cabrucas* between Camacan and Itabuna are considered the main area of occurrence of the acrobat bird (*Acrobatornis fonsecai*), a species that was first discovered and described in a *cabruca* (Pacheco *et al.* 1996). Most of the subsequent recordings of the species are also linked to the *cabruca* environment and Pacheco *et al.* (1996) believe that the conversion of natural forest into *cabrucas* may have actually contributed to a widening of its range, including to altitudes of up to 600 m. The acrobat bird is a canopy species and observations have indicated that it forages in mixed flocks, feeding mainly on insects and nesting preferentially in legume trees which are common in *cabrucas*. The species has recently been reported from *cabrucas* near the Una Biological Reserve (André de Luca, pers. com. 2007) and in the Ilhéus area (Faria *et al.* 2006).

Invertebrates: Cabrucas may also play a role in invertebrate conservation. The ant *Dinoponera lucida* is on the list of Brazilian fauna threatened with extinction. It is found in the southern part of the cocoa region, including in cocoa plantations adjacent to forest

remnants, indicating that, as a minimum, *cabrucas* can facilitate species dispersion among forest remnants (Delabie *et al.* 2007). On the other hand, the rare endemic ant species, *Blepharidatta* sp. n., has been found in forest areas through much of the cocoa region, but only a single encounter in a *cabruca* has so far been reported (Figure 1). These ants are exceptional by not having winged forms and thus depend on suitable habitat for their dispersal. If it is true that *cabrucas* do not provide suitable habitat, then these ants would be severely threatened by forest fragmentation (J.H.C. Delabie, unpublished data).

While several endangered species of mammals, birds and invertebrates of the region use *cabrucas* as (secondary) habitat, for others these agroforests do not seem to offer suitable conditions. Two endangered primates of the region, the yellow-breasted capuchin monkey (*Cebus xanthosternos*) and the Southern Bahian masked titi monkey (*Callicebus melanochir*), as well as the bristle-spined porcupine (*Chaetomys subspinosus*), do not seem to enter *cabrucas* (Alves 1990, Giné *et al.* 2006, Neves *et al.* 2007), reinforcing again the complementarity of forest conservation and the conservation of *cabrucas* in an integrated landscape management strategy. Still, *cabrucas* may benefit these species indirectly by helping to maintain vital ecological processes, including maintaining prey species and reducing edge effects in forest fragments.

1.4. Discussion and recommendations

Landscape management in southern Bahia cocoa region

Research in the cocoa region of southern Bahia has shown that although *cabrucas* are sub-optimal habitat for many typical forest species, they can to some extent mitigate the fragmentation and loss of natural habitat for a significant part of the regional biota. In the forest-rich landscape of Una, small forest remnants (< 200 ha) have biological communities as rich and abundant as the large forest blocks, suggesting that the *cabrucas* pose no obstacle to the movements of forest species through the landscape. In the forest-poor landscape of Ilhéus, on the other hand, the *cabrucas* that dominate the landscape provide habitat for many species while the remaining forest fragments are essential for those species that cannot use *cabrucas*. Large forest remnants are then fundamental to maintaining viable populations, especially for rare, large-bodied and area-demanding species. Research in other regions has shown that species that tolerate matrix habitat (such as *cabrucas*) are better able to survive in fragmented forest landscapes than species that do

not, with the latter being particularly dependent on large, contiguous forest blocks (Laurance 1991, Gascon *et al.* 1999), hence the importance of an adequate system of protected areas and private forest reserves even in an overall "biodiversity-friendly" landscape like the cocoa region of southern Bahia.

Theoretical studies on forest fragmentation suggest that below 20 to 30% forest remnants in the landscape, species richness is negatively affected by fragmentation over and above forest loss *per se* (Fahrig 1998). While the lower overall diversity of the Ilhéus landscape (with 5% forest) compared to the Una landscape (with 50% forest) is in agreement with this theoretical conclusion, it is nevertheless striking that the forest-poor landscape of Ilhéus still has relatively diverse biological communities with presence of a large proportion of the native species of the region. There is no doubt that the combination of forest remnants with a matrix dominated by *cabrucas* and secondary forests are fundamental for the maintenance of these rich biological communities. Further studies are needed, however, to define limits in terms of percent cover and spatial configuration of forest remnants as well as the percent cover, configuration and characteristics of the *cabrucas* and other matrix habitat beyond which ecological processes and biodiversity would be substantially and irreversibly impaired.

While such questions await further study, some lessons clearly emerge from the existing evidence. In cocoa landscapes with still a high proportion of forest remnants (as in Una), these should be conserved to ensure the perpetuation of forest species, while *cabrucas* should be maintained in the matrix especially to provide connectivity and prevent the impoverishment of isolated, small remnants. In landscapes dominated by *cabrucas* with only little remaining forest cover (as in Ilhéus), on the other hand, the forest area should be increased to ensure the conservation of strict forest species, while the *cabrucas* should be maintained both for their essential role as habitat for many native species and to ensure a maximum of connectivity among the forest remnants. In this latter situation, adherence to current legislation requiring legal reserves (20% of each property) and areas of permanent preservation (river margins and hill slopes) of native vegetation would lead to a substantial increase of total forest cover to about 30%, in addition to the public protected areas (Faria 2007). The creation of private reserves (for which Brazilian law offers tax rebates) and expansion of public protected areas are further options to increase the total amount of forest habitat. Care should be taken to increase the proportion of natural forest in the landscape in a way that takes account of the beta diversity of natural species assemblages

and is adequately stratified across underlying climatic, edaphic and topographic features. Special attention needs to be given to species with limited capacity to use *cabrucas* (Table 2). Action plans should be developed to maintain viable populations of these species in large remnants, possibly aided by dispersion corridors whose effectiveness should be monitored.

While all conversion of mature forest is illegal in the Atlantic Forest biome, the clearing of fallow land (e.g. for pasture or annual crops) should be done in such a way that forest remnants and *cabrucas* do not become isolated and that abrupt borders between forest and open areas are not unnecessarily created. Microclimatic effects and especially the increased prevalence of fire tend to progressively degrade especially small forest fragments with exposed borders (Gascon *et al.* 2000). Such effects can probably be reduced if forest edges are buffered by fallows, secondary forest or *cabrucas*, although the effectiveness of such spatial arrangements has not been established in Bahia.

Cocoa management within the cabruca *system*

The conservation of diversified and structurally complex shade canopies composed of native trees species in the *cabrucas* is important for the conservation of a large number of native tree species and as a structurally diverse habitat for many other organisms. Research is needed to establish new recommendations for shade tree densities, taking into account not only direct effects of shade on cocoa production, but also effects on production costs via suppression of weeds and biological pest control, as well as tree products (fruits, wood etc.). Incipient markets that specifically reward the conservation-friendly production of cocoa in *cabruca* systems may expand in the future. The internalization of environmental services such as carbon sequestration and contribution to the scenic beauty of the southern Bahian landscape with its high potential for tourism would create further incentives for maintaining traditional shade canopies. Guidelines need to be developed for managing the regeneration of the shade trees in *cabrucas* to prevent the gradual loss of late successional tree species and their replacement by (often exotic) pioneers. The management should ensure the persistence of endemic and endangered tree species (Figure 4), as well as species used by endemic fauna (Table 3), and should strengthen those features of *cabrucas* that are known to contribute to their habitat quality. Hollow trees, fallen logs, termite nests, trees and branches supporting bird's nests (especially of rare species) and other structures

that are used by fauna should be conserved. Epiphytes and vines should not be removed from shade trees unless they interfere with the management of the area. When thinning or pruning the canopy, it may often be possible to maintain some connectivity in the canopy to facilitate the movement of species such as sloths and other arboreal fauna, especially where *cabrucas* abut on natural habitat. Education and capacity building programs that increase cocoa farmers' ability to identify tree species and their seedlings in the field are also needed (Eduardo Mariano Neto, pers. com. 2008). Pesticides should be used only locally if at all, and preference should be given to organic practices.

Hunting is illegal throughout the Atlantic Forest biome, although this law is generally not enforced. Hunting is widespread in the region and is likely to affect the density and distribution of most medium to large terrestrial mammals (Pardini 2001). Observations suggest that the farm owners' prohibition of hunting and their encouragement to keep small livestock as an alternative source of protein have potential to reduce hunting by farm workers (Santos 1999). Snakes are often killed for fear and because farm workers are often unable to distinguish poisonous from nonpoisonous species, hence the need for educational programs about the prevention of snake accidents, the harmlessness of most snake species, and the ecological importance of snakes in the control of their prey (e.g. rodents some of which can damage cocoa; Argôlo 2004). These recommendations are broadly valid for shaded cocoa (and other tree crop) systems elsewhere (Schroth *et al.* 2004).

Conservation of endangered species

Cabrucas should be considered priority landscape elements in action plans for the conservation of the pileated antwren (*Herpsilochmus pileatus*), the acrobat bird (*Acrobatornis fonsecai*), the golden-headed lion-tamarin (*Leontopithecus chrysomelas*), the maned three-toed sloth (*Bradypus torquatus*), the painted tree rat (*Callistomys pictus*) and the ant *Dinoponera lucida*. However, with the possible exception of the acrobat bird, sufficiently large forest remnants are essential for the conservation of these species, in addition to *cabrucas*. More research is needed to establish the exact role that *cabrucas* can play in the life cycles of these species and to identify management practices to increase their habitat value. Educational programs to increase the knowledge about these rare and endangered species in *cabrucas* among farm owners and workers should also be a priority.

1.5. Conclusions

This review demonstrates that *cabrucas* are utilized by a substantial part of the fauna and flora native to southern Bahia. Given the high degree of forest loss and fragmentation in parts of the region, *cabrucas* have an important role to play in biodiversity conservation by providing alternative or additional habitat for many forest species, increasing connectivity between forest fragments, and reducing edge effects to which fragments are exposed.

Conservation strategies should consider the role of *cabrucas* at two spatial scales: the local management of the plantations, and the scale of the landscape within which *cabrucas* interact with other vegetation types. The studies reviewed here indicate that where a large part of the landscape is still covered by forest remnants, the landscape as a whole, and the *cabrucas* within it, are richer in species than where much of the original forest cover has been converted into *cabrucas* and other land-uses. Initiatives to conserve *cabrucas* as important habitat for many native species should thus always be seen in context with the conservation and restoration of natural forest habitat. Brazilian environmental legislation offers a legal framework to increase forest cover to 20-30% outside of protected areas, a value that, according to theoretical models, would be enough to support metapopulation dynamics in a fragmented landscape, especially where much of the remainder is covered by *cabrucas* that are sympathetically managed for conservation. Current scientific knowledge of the ecology of many species of the cocoa producing region of Bahia already allows us to identify management practices that help to maintain and increase the habitat value of *cabrucas*, although studies are needed on questions such as optimum shade tree densities and management practices to ensure the regeneration of canopy trees. Since *cabrucas* are production systems, any recommendation needs to take economic viability into account, and both costs and benefits of conservation friendly management should be monitored to inform a long-term plan of adaptive management. Long-term scientific studies are needed that relate biological diversity and the persistence of sensitive species to different combinations of *cabrucas* and natural forest in landscapes in order to determine threshold levels of deforestation and fragmentation beyond which biodiversity is significantly impaired. Further research should also be encouraged to relate variables such as the presence of food resources, shelter, vegetation structure and microclimate to the richness and composition of native species assemblages and especially the presence of sensitive forest species in *cabrucas*. It should also be remembered that no two *cabrucas* are the same. *Cabrucas* are derived from already highly heterogeneous forest through histories

of establishment and subsequent plantation management that are, in combination, unique for each area. Thus, even when a list of general recommendations is set out to reconcile production and conservation, adaptation to local conditions will always be necessary, requiring ecologically educated and conscious farm owners and workers.

Finally, while this review focused on Bahia with its specific fauna, flora and culture of cocoa growing, many of our conclusions on the management of landscapes and cocoa production systems to reconcile biodiversity conservation with tree crop production are applicable to other geographical regions and add to an increasing pool of information on the biodiversity-friendly management of tropical land-use mosaics (Schroth *et al.* 2004, Schroth & Harvey 2007).

Acknowledgments

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Table 1: Number of species and total recordings of small mammals, bats, birds, litter frogs and lizards, frugivorous butterflies, ferns, and bromeliads in the interior of large forest remnants and *cabrucas* in the county of Una, southern Bahia, Brazil.

Biological group	Forest interior	*Cabruca*	Species common to both habitats	Principal changes in communities	Source
Small mammals				Change in the relative dominance among species	Pardini 2004
# recordings	195	299			
# species	11	13	*		
Bats				Greater richness and abundance in *cabrucas* (with possible influence of the sampling method in the more open environment); change in the relative dominance between the two most common species	Faria 2002; Faria *et al.* 2006
# recordings	278	1314			
# species	18	39	16		
Birds				Greater richness in *cabrucas* as effect of the invasion of species from more open environments; change in the relative abundance among species	Laps 2006
# recordings	1610	2459			
# species	121	158	94		
Litter frogs				No significant change in richness, abundance or species composition	Dixo 2001
# recordings	763	217			
# species	12	12	11		
Litter lizards				Change in the relative dominance among species	Dixo 2001
# recordings	53	148			
# species	10	9	9		
Frugivorous butterflies				Change in the relative dominance among species	Accacio 2004
# recordings	233	636			
# species	43	51	30		
Ferns				Different species composition	Paciencia & Prado 2005
# recordings	1479	1019			
# species	32	26	7		
Bromeliads				Different species composition	Alves 2005
# recordings	711	264			
# species **	25	18	13		

*data not available; **species or morpho-species

Table 2: Species known to rarely occur in *cabrucas*

Taxon	Common name	Restrictions to use of *cabrucas*	Source
Mammals			
Cebus xanthosternos	yellow-breasted capuchin monkey		Alves 1990;
Callicebus melanochir	coastal black-handed titi		
Mazama sp	deer	Low probability to use *cabrucas*	Giné *et al.* 2006
Pecari tajacu	wild pig		
Chaetomys subespinosus	bristle-spined porcupine		
Callistomys pictus	painted tree rat	Relatively tolerant to the habitat modifications occurring when forest is converted into *cabruca*, but abundance in these plantations probably reduced by hunting	Moura 2005; Moura 2008;
Didelphis marsupialis	common oppossum		Alves 1990
Bradypus torquatus	maned three-toed sloth	Use of *cabrucas* dependent on the occurrence of plant species that make up its diet, density of shade trees and proximity of forest remnants	Cassano 2006; Raboy *et al.* 2004;
Leontopithecus chrysomelas	golden-headed lion-tamarin		Alves 1990
Oryzomys laticeps	forest rat		
Marmosops incanus		Low probability of using *cabrucas*	Pardini 2004
Monodelphis americana	three-striped short-tailed oppossum		
Rhinophylla pumilio	bat	Low probability of using *cabrucas* when they are inserted in a landscape with little native forest	Faria *et al.* 2006; Faria & Baumgarten 2007
Birds			
Lipaugus vociferans		Low probability of using *cabrucas*	Faria *et al.* 2006
Drymorphyla squamata			
Herpsilochmus pileatus	"chorozinho-da-Bahia"	Low probability of using *cabrucas* when they are inserted in a landscape with little native forest	Faria *et al.* 2006
Pyriglena leucoptera			
Rhytipterna simplex			
Formicarius colma			
Formicaridae and Pipridae	understory birds	Low probability of using *cabrucas*	Alves 1990
Cracidae and Tinamidae	terrestrial birds		

Table 2 (cont.)

Taxon	Common name	Restrictions to use of *cabrucas*	Source
Reptiles and Amphibians			
Enyalus catenatus pictus	lizard		
Adelophrine pachydactyla	litter frog		Dixo 2001;
Cycloramphus migueli	litter frog	Low probability of using *cabrucas*	
Leptodactylus spixi	litter frog		Faria *et al.* 2007
Chiasmocleis gnoma	littler frog		
Bothrops bilineatus	forest pitviper	Low probability of using *cabrucas*	Argolo 2005
Lachesis muta	South American bushmaster		
Invertebrates			
Dinoponera lucida	ant		
Typhlomyrmex spp.	ant	Low probability of using *cabrucas*	Delabie *et al.* 2007
Family Ecitoninae	army ants		
Subfamily Morfidae	fruit eating butterflies	Low probability of using *cabrucas*	Accacio 2004
Epiphytic bromeliads			
Aechmea mollis			
A. turbinocalyx			
Areococcus paviflorus			
Lymania globosa			
L. azurea	understory bromeliads	Low probability of occurring in *cabrucas*	Alves 2005
L. smithii			
Nidularium amorimii			
Vriesea drepanocarpa			
V. duvaliana			
Aechmea conifera			
A. leonard-kentiana	canopy bromeliads	Low probability of occurring in *cabrucas*	Alves 2005
Aechmea sp.			

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Table 3: Tree species recorded in *cabrucas* (Sambuichi and Haridasan 2007) that are part of the diet of golden-headed lion-tamarins - GHLT (Raboy *et al.* 2004) and maned three-toed sloths (Cassano 2006, Correia *et al.* 2006) or listed as endemic (Thomas 2003) or endangered species (IUCN 2008).

Family / *Species*	Common name	GHLT diet	Sloth diet	Endemic	IUCN[1]
Anacardiaceae					
Tapirira guianensis	pau-pombo	x	x		
Annonaceae					
Rollinia bahiensis				x	VU
Apocynaceae					
Lacmellea aculeata		x			
Rauvolfia bahiensis				x	
Bignoniaceae					
Tabebuia elliptica		x			
Bombacaceae					
Eriotheca sp.	imbiruçú		*E. globosa*		
Caesalpiniaceae					
Arapatiella psilophylla				x	VU
Caesalpinia echinata	pau-brasil				EN
Dialium guianense	jitaí-preto	x	x		
Hymenaea oblongifolia				x	
Macrolobium latifolium		x	x		
Senna multijuga	canafístula/cobi		x		
Cecropiaceae					
Cecropia spp.	embaúba		*Cecropia*		
Clusiaceae					
Symphonia globulifera	guanandí	x			
Ebenaceae					
Diospyros melinonii		x			
Euphorbiaceae					
Mabea piriri		x			
Fabaceae					
Dalbergia nigra	jacarandá				VU
Lauraceae					
Aniba intermedia					VU
Lecythidaceae					
Cariniana legalis	jequitiba-cipó				VU
Lecythis lurida	inhaíba				LR/cd
Malpighiaceae					
Byrsonima laevigata		x			

Table 3 (cont.)

Family / Species	Common name	GHLT diet	Sloth diet	Endemic	IUCN[a]
Meliaceae					
Cedrela odorata	cedro roxo				VU
Mimosaceae					
Parkia pendula	jueirana	x			
Monimiaceae					
Bracteanthus atlanticus				x	
Moraceae					
Artocarpus heterophyllus	jaqueira	x			
Brosimum guianense			x		
Brosimum rubescens	condurú	x	x		
Ficus pulchella	figueira				VU
Ficus spp.	figueira	x	x		
Sorocea guilleminiana					VU
Myristicaceae					
Virola gardneri	bicuíba		x		
Myrtaceae					
Eugenia flamingensis				x	
Eugenia sp.		x			
Psidium sp.		*P. guajava*			
Polygonaceae					
Coccoloba alnifolia		*Coccoloba*			
Rubiaceae					
Guettarda platyphylla		x			
Santalaceae					
Acanthosyris paulo–alvinii				x	
Sapotaceae					
Chrysophyllum splendens					VU
Manilkara elata	maçaranduba	*Manilkara*			EN
Micropholis compta					VU
Micropholis					LR/cd
Micropholis sp.		*Micropholis*			
Pouteria beaurepairei					LR/cd
Pouteria grandiflora					LR/nt
Pouteria bangii		x			
Simaroubaceae					
Simarouba amara	arubá	x			

[a] LR = Lower risk - species assessed that do not fit into threat categories (cd = taxon whose conservation is ensured by a specific conservation program; nt = near threatened); VU = vulnerable; EN = endangered.
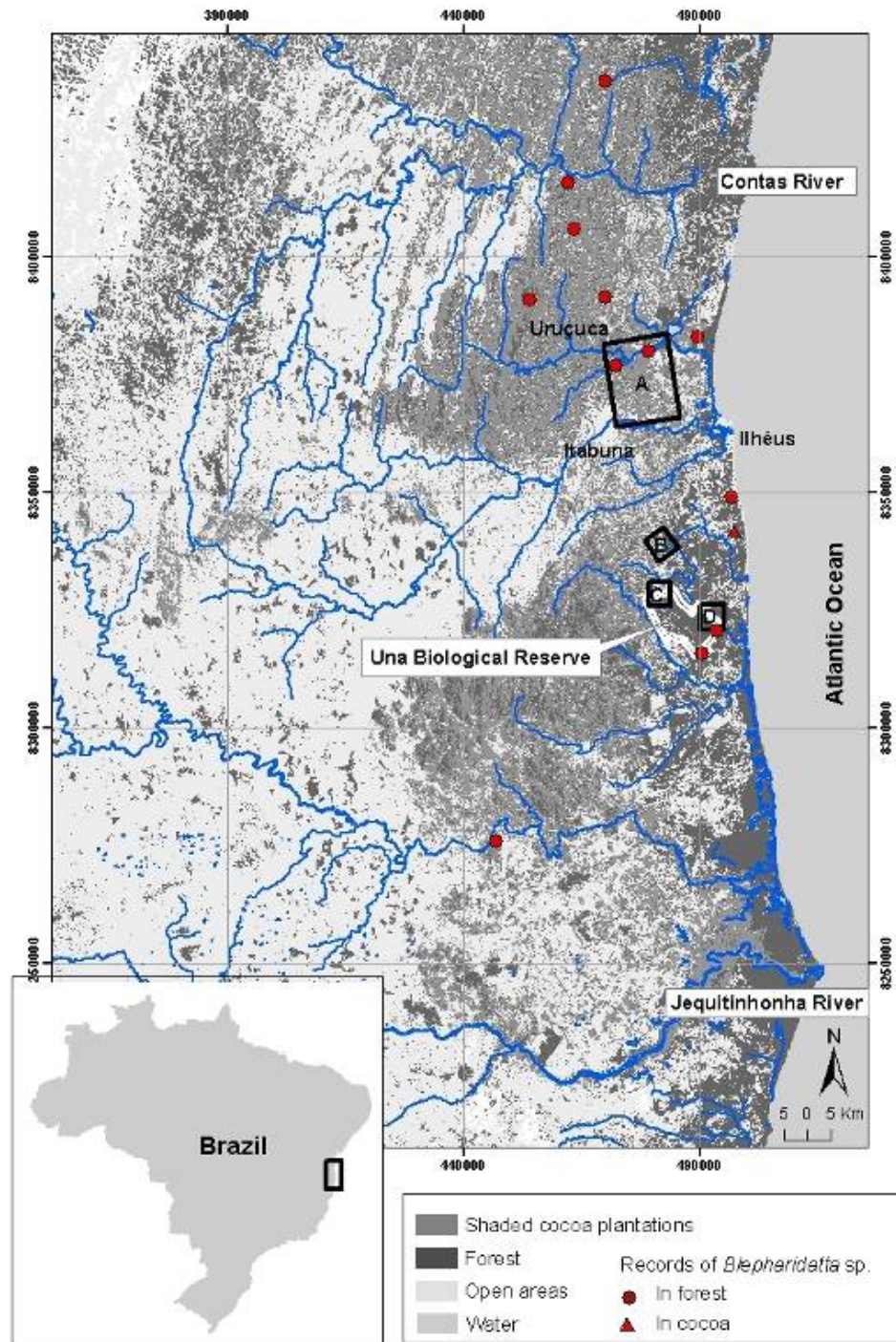
Figure 1: The cocoa region of southern Bahia, Brazil, delimited by the Rio Contas and Rio Jequitinhonha, with the location of two landscapes with low forest cover (Ilhéus – A) and high forest cover (Una - B, C and D) where the biodiversity of *cabrucas* and forests was studied. The symbols refer to sampling locations of the endemic ant species, *Blepharidatta* sp. n (modified after Landau et al. 2003).

(A)

(B)

Figure 2: Traditional *cabrucas* of southern Bahia, Brazil, retain much of the original forest structure. A) *Cabruca* in the county of Una; B) Aerial view of a *cabruca* in the same region (Photos by G. Accacio, with permission).
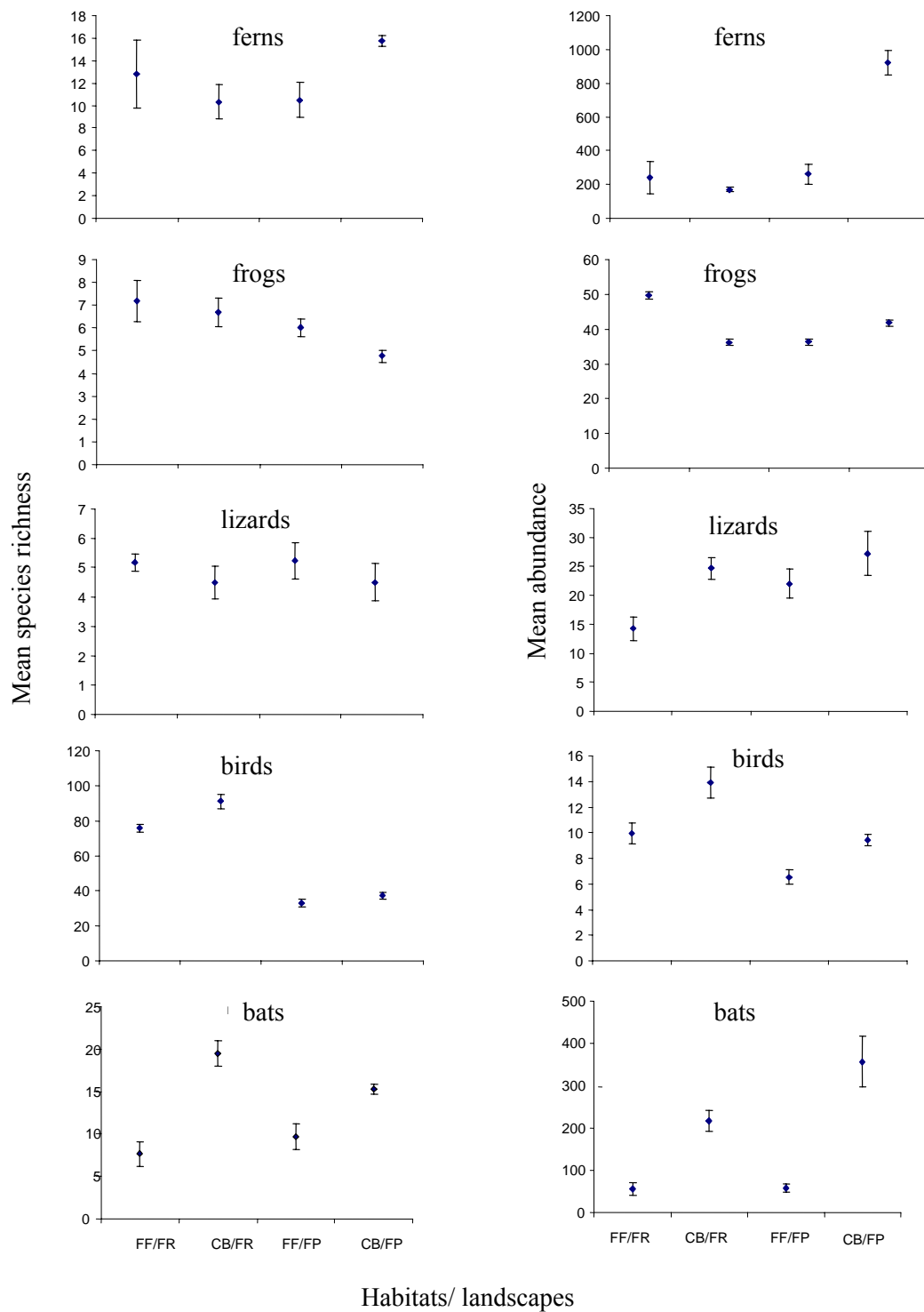
Figure 3: Richness and abundance of different biological groups in forest fragments (FF) and *cabrucas* (CB) in a forest-rich (Una - FR) and a forest-poor (Ilhéus - FP) landscape in southern Bahia, Brazil (from Faria *et al*. 2007, modified).

Figure 4: Brazil wood tree (*Caesalpinia echinata*) with high density of epiphytes in a *cabruca* in the region of Ilhéus where *cabrucas* represent most of the tree cover in the landscape (photo by G. Schroth).

Large mammals in an agroforestry mosaic in the Brazilian Atlantic forest

Camila Righetto Cassano, Jos Barlow and Renata Pardini

Abstract

The forest-like characteristics of agroforestry systems create a unique opportunity to combine agricultural production with biodiversity conservation in human-modified tropical landscapes. The cacao growing region in southern Bahia, Brazil, encompasses Atlantic forest remnants and large extensions of agroforests, locally known as *cabrucas,* and harbors several endemic large mammals. Based on the differences between *cabrucas* and forests, we hypothesized that: (1) non-native and carnivore mammals that feed on small prey are more frequent, while arboreal, frugivore and hunted mammals are less frequent, in *cabrucas* than forests; (2) the two systems differ in assemblage structure and composition, but not in species richness; and (3) higher species turn-over occurs in *cabrucas*. We then used camera-traps to sample mammals in nine pairs of *cabruca*-forest sites. The high conservation value of the agroforests was supported by the presence of species of conservation concern in *cabrucas*, and similar species composition between forests and *cabrucas*. However, arboreal species were less frequently recorded, and non-native species and a carnivore adapted to open environments were more frequently recorded, in *cabrucas*. Additional factors that may overestimate the conservation value of *cabrucas* are: the high proportion of total forest cover in the study landscape, the apparent impoverishment of large mammal fauna in forest controls, and the uncertainty about long term maintenance of agroforestry systems. We highlight the importance of agroforests and forest remnants to provide connectivity in deforested tropical forest landscapes, but emphasize the importance of controlling hunting and dogs in order to increase the value of agroforestry mosaics.

Key words: Camera-trap; fragmentation; matrix; shade cacao plantation; *Theobroma cacao*; vertebrates

2.1. Introduction

Protected areas have been created with the main purpose of biodiversity conservation. Nevertheless, they alone do not guarantee the maintenance of ecological processes over large spatial and temporal scales, and are dependent on the long term processes and dynamics of the surrounding landscape (Rosenzweig 2003, Rodrigues *et al.* 2004, Chazdon *et al.* 2009, Gardner *et al.* 2009). Human-modified landscapes are composed of a mosaic of environments with different degrees of suitability for the occurrence and dispersal of individuals from native and non-native species (Daily *et al.* 2003, Harvey *et al.* 2006, Harvey & Villalobos 2007, Umetsu *et al.* 2008). Understanding changes in populations and communities between different environments and the linkages and dynamics across landscape mosaics are research priorities for conservation planning of tropical forests (Chazdon *et al.* 2009, Gardner *et al.* 2009).

In tropical forest regions, agroforestry systems represent an opportunity for the development of multi-functional landscapes that combine agricultural production and biodiversity conservation as they retain more forest-like characteristics than any other agricultural land-uses (Perfecto & Vandermeer 2008). Landscapes harboring large extensions of cacao (*Theobroma cacao*) and coffee (*Coffea arabica*) agroforests have raised special conservation interest, since they occur in areas originally occupied by biologically-diverse tropical forest (Rice & Greenberg 2000, Clough *et al.* 2009, Tscharntke *et al.* 2011). Species richness and abundance in cacao and coffee agroforests may parallel those of primary forest, but the composition and structure of communities usually differ. As a general rule, these shaded plantations harbor more generalist and fewer forest-dwelling species of small vertebrates and invertebrates than forests (Pardini *et al.* 2009, Waltert *et al.* 2011). However, species richness and composition in agroforests depends on the maintenance of forest cover at larger spatial scales, as has been shown in studies contrasting landscapes with different proportions of forest cover (Faria *et al.* 2007) or plantations at varying distances to the nearest forest (Bali *et al.* 2007). Moreover, land-use intensification - decreasing shade and shade diversity, and increasing agrochemical use - are negatively correlated with species richness (Perfecto *et al.* 2005, Gordon *et al.* 2007).

Large mammals play important roles in many ecosystems through a diverse array of mechanisms. These roles can be either played directly, via seed dispersal and predation (Tabarelli & Peres 2002, Terborgh *et al.* 2008) or prey population control (Terborgh *et al.* 2001, Sinclair *et al.* 2003), or indirectly, by helping maintain assemblages of dung-beetles

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responsible for nutrient cycling, soil aeration and the secondary dispersal of seeds contained in mammal dung (Nichols *et al.* 2009). Many large mammals are threatened by habitat loss and disturbance, and overhunting is also a significant threat (Morrison *et al.* 2007) that can extirpate species from apparently intact forests (Redford 1992). The ease of access to forest remnants in highly fragmented landscapes increases both hunting pressure (Chiarello 1999, Cullen *et al.* 2000, Peres & Nascimento 2006, Michalski & Peres 2007, Galetti *et al.* 2009) and the damage caused by invasive species (e.g. domestic dogs – Fiorello *et al.* 2006). Furthermore, human-wildlife conflicts resulting from crop damage and predation of livestock often result in the persecution and death of mammals (Naughton-Treves *et al.* 2003, Michalski *et al.* 2006a). Studies of large mammal communities in agroforestry systems have shown that species differ in their sensibility to the conversion of forest into agroforests, but species traits driving such responses are still poorly understood (Estrada *et al.* 1994, Harvey *et al.* 2006).

The Brazilian Atlantic forest has endured a long history of human influence, and current strategies for biodiversity conservation in this biome require the reestablishment of functional connectivity between forest remnants (Tabarelli *et al.* 2005, Ribeiro *et al.* 2009). The Atlantic forest from southeastern Bahia is recognized for its high species richness and endemism (Silva & Casteleti 2003, Martini *et al.* 2007). It also encompasses large extensions of agroforests, with approximately 6000 km^2 of land under cocoa plantations shaded by native trees (data from the years 1996/97; Landau *et al.* 2008). These agroforests are known regionally as *cabrucas*, and their management is likely to be of major importance for biodiversity conservation (Cassano *et al.* 2009).

Like other agroforestry systems, *cabrucas* are structurally simplified when compared to native forests; the canopy layer has lower tree diversity and density, the understory is exclusively composed by cacao trees and undergrowth vegetation is periodically removed by mechanical and chemical means (Sambuichi 2002). This structurally simplified vegetation has negative effects on species that disperse and forage through the vegetation, such as small arboreal rodents and marsupials, and understory birds (Moura 1999, Faria *et al.* 2006). The same negative effect can be expected on larger arboreal mammals as well as on frugivores, since this simplified vegetation should present lower fruit productivity. On the other hand, large mammals that feed on small vertebrates and insects might find higher food availability in *cabrucas* than in forests. The abundance of such prey is similar in *cabrucas* and forests (Faria *et al.* 2007, Pardini *et al.* 2009), but may be easier to capture in

the more open *cabrucas*. Other differences in large mammal assemblages between *cabrucas* and forests are expected to result from the increased human and invasive species disturbance in managed agroforest compared to forest. Finally, there is also great variation in the implementation processes and management practices (Sambuichi & Haridasan 2007) that might produce greater among-site variation in resources and disturbances, and thus higher beta diversity, within *cabrucas* than within forests.

Few studies have been conducted on the use of *cabrucas* by large mammals, and these focused on single, endangered species (Raboy *et al.* 2004, 2008; Cassano *et al.* 2011, Oliveira *et al.* 2011). We used camera-traps and a paired design of *cabrucas* and forest remnants to sample large mammals across a complex agroforestry mosaic in southern Bahia, and test the following hypotheses: (1) non-native and carnivore mammals that feed on small prey use *cabrucas* more frequently, while arboreal, frugivore and hunted mammals use these systems less frequently than forests; (2) consequently, the two systems differ in assemblage structure and composition, but not in richness (alpha diversity); and (3) species turn-over (beta diversity) is higher in *cabrucas*. We use our results to critically examine the conservation value of *cabrucas*.

2.2. Methods

Study area and sampling design

This study was carried out in the cacao growing region of southern Bahia, in a landscape encompassing part of the municipalities of Una and Arataca (~60 km^2; between 39°5'- 39°22' W and 15°4'- 15°14' S), largely covered by mature and late secondary forest (roughly 50% of the area; see Figure 1). The original vegetation is the Southern Bahian Wet Forest (Mori *et al.* 1983, Thomas 2003), which varies among lowland rainforest over latisols, lowland rainforest over podzols and montane forest (above the 500 m asl) from East to West in the study region (Thomas 2003). The mean annual temperature is 24°C and rainfall is around 2000 mm/year. There is no clear seasonality, although a warmer and rainless period usually occurs between December and March (Mori *et al.* 1983, Oliveira & Fontes 2000).

The study sites were located in farms around and between two protected areas: the Una Biological Reserve in the East, and the Serra das Lontras National Park in the West. This agroforestry mosaic outside the reserves (Figure 1) is covered by 38 percent of mature and

late secondary forests, and by 21, 20 and 10 percent of *cabruca*, pasture and permanent monocultures respectively. The land-use is strongly related to edaphic and topographic conditions, and larger forest patches persist in areas that are less favorable for the development of cacao plantations (sandy soils in the East and the steeper slopes in the West).

We sampled 18 sites located in nine blocks, each composed of a *cabruca* site and an adjacent site in a forest remnant (varying from tens to hundreds of hectares in size) 200 to 450 m apart from each other. The criteria for site selection were: (1) cacao plantations shaded mainly by native trees (excluding the more intense managed plantations) and adjacent to a forest remnant (used as a control); and (2) blocks were at least 2.5 km from each other (distances varied from 2.5 to 6.2 km). The block design increased the power of our analysis by helping control for the variable landscape context, soil, and topography.

Large mammal sampling

Two camera-traps (analog Trapacamera – www.trapacamera.com) were placed in each site, one on the ground and one in the understory (3-4 m above ground level), 50 to 100 m apart from each other. Ground stations were baited with banana and sardine, and understory station just with banana. The position of cameras and the types of bait aimed at being able to sample terrestrial as well as arboreal, frugivoure as well as carnivore mammals. Camera-traps are frequently used to sample terrestrial large mammals (Silveira *et al.* 2003, Tobler *et al.* 2008a, Espartosa et al., in press) and have also proved effective for sampling arboreal species (Kierulff *et al.* 2004).

Data collection occurred during four sampling sessions in July-October of 2007 and 2008, and January-April of 2008 and 2009, when all sites were sampled. During each of the four sampling sessions, three *cabruca*-forest pairs were sampled simultaneously for four consecutive weeks and checked weekly to exchange film and re-bait (all nine pairs of sites sampled within four months). Nevertheless, the malfunction of camera-traps and the depletion of some films led to an unequal effort among sites and stations, which ranged from 90 to 127 trap day in forest sites and from 100 to 128 trap day in *cabruca* sites. The total sampling effort was 1895 and 2080 trap day, respectively.

Species classification and expected responses to agroforest

The potential pool of 32 native large mammals occurring in the study region was identified following Moura (2003), excluding *Pseudalopex vetulus* which is not considered native to the region (Emmons & Feer 1999). Species were classified as hunted or not following Cullen *et al.* (2000), Peres (2001), Naughton-Treves *et al.* (2003), Michalski *et al.* (2006a), Peres & Nascimento (2006) and Whiteman *et al.* (2007). Hunted species included ungulates, armadillos, large primates and large rodents hunted for meat, and felids, which are often persecuted to protect livestock. We also included the opossum (*Didelphis aurita*) as it is known to be heavily hunted in the study region (Santos 1999). Classification of species according to dietary and locomotor habits followed Fonseca *et al.* (1996). Dietary classes were simplified to "frugivore", including all species that rely mainly on plant material with large proportion of fruit (frugivore/omnivore plus frugivore/herbivore) and "carnivore", including all species that include prey in their diet (omnivore plus carnivore).

From the 32 species, 14 were classified as hunted, 10 as arboreal, seven as frugivore and 19 as carnivore (Table 1). All species classified as frugivore were also either hunted or arboreal. Three species (*Sylvilagus brasiliensis, Tamandua tetradactila* and *Lontra longicaudis*) were not classified, but these were also rare or absent in our data and were not suitable for further analyses. Based on these classifications and our hypotheses, we expected 24 species to exhibit lower capture rates in *cabrucas* than forests and five to exhibit the opposite trend (Table 1).

Data analysis

The capture rate was calculated for each species and site by dividing the total number of records by the total sampling effort. A single record was defined as the presence of at least one picture of one species in one station during a 24-h period (starting at 00 00 h ending at 24 00 h). To calculate sampling effort (trap-days) we only included days with no apparent camera-trap malfunction or film depletion. For arboreal and scansorial species we considered both ground-level and understory stations. We calculated capture rate by strata and then the average between them to prevent uneven effort between stations to bias capture rate. For terrestrial species, just the ground-level stations were considered. To compare the capture rate of each species between forests and *cabrucas*, we used a permutation test equivalent to a paired t-test, with 10,000 randomizations implemented in

R 2.10.1 (R Development Core Team 2009). Only species recorded in at least four sites in one type of system (forests or *cabrucas*) were included in this analysis.

We used non-metric multi-dimensional scaling (MDS) and analysis of similarity (ANOSIM) to summarize and test for differences in assemblage structure and composition between forests and *cabrucas*. Analyses were run using square-root transformed capture rates (for structure) or presence-absence data (for composition) and Bray-Curtis distance. A similarity percentage analysis (SIMPER) was used to identify the contribution of each species to the dissimilarity in assemblage structure between forests and *cabrucas* (Clarke 1993). To test for differences in species turn-over (beta-diversity) between forests and *cabrucas*, we used an analysis of multivariate homogeneity of group dispersions and a permutation test (10,000 randomizations), which tests for differences in assemblage composition among groups contrasting the average dissimilarity from individual sites to their group centroid in multivariate space (Anderson *et al.* 2006). The MDS, ANOSIM and multivariate homogeneity of group dispersions analyses were implemented in R 2.10.1 (R Development Core Team 2009), using the Vegan package. SIMPER analysis was implemented in PRIMER 5.2.4 (Clarke & Gorley 2006).

Alpha diversity was estimated for each site using Jackknife1 (as suggested by Tobler *et al.* 2008a,b) using sampling day as the unit of effort, and considering only the days in which both camera-traps (ground and understory stations) were working (78 to 126 days among sites). The abundance-based coverage (ACE) and the Chao1 estimators were also calculated and produced similar results, except for two *cabruca* sites with large number of rare species where alpha diversity estimated by ACE was 1.5 times larger than that found by other estimators. We compared differences in alpha diversity between forests and *cabrucas* using the same randomization procedure as previously described for species capture rates, and considering native species only. To examine gamma diversity, we constructed species accumulation curves with 95% CI for each environment, considering sites as the unit of effort. To standardize sampling effort per site we only used records of native species from the first 78 days in which camera-traps in both stations per site worked simultaneously. Mean number of species and the 95% CI were calculated by the Mao Tau method, using EstimateS 8.2.0 (Cowell 2006).

2.3. Results

We recorded 19 native and three non-native large mammal species in trap stations. Fifteen native and three non-native species were registered in forest remnants (1010 and 22 records, respectively), while 17 native and three non-native were registered in *cabrucas* (460 and 76 records, respectively; Table 1). Capture rates were higher but fewer species were recorded in understory stations. Four additional species that eluded the camera-trap stations were fortuitously detected once or twice in *cabrucas* (*Bradypus torquatus*, *Pecari tajacu* and *Tamandua tetradactyla*) and forest remnants (*Callicebus melanochir*) during the period of this study.

Capture rates in forests and cabrucas

Opossum (*Didelphis aurita*), black-tufted-ear marmoset (*Callithrix kuhlii*), golden-headed lion-tamarin (*Leontopithecus chrysomelas*), nine-banded armadillo (*Dasypus novemcinctus*), coati (*Nasua nasua*), tayra (*Eira barbara*), crab-eating raccoon (*Procyon cancrivorus*), crab-eating fox (*Cerdocyon thous*) and domestic cod (*Canis familiaris*) were recorded in at least four forest or *cabruca* sites. Our expectation that capture rate would be lower in *cabrucas* was met for both arboreal species (*C. kuhlii* and *L. chrysomelas*; Figure 2A-B). Our expectation that capture rate would be higher in *cabrucas* was met for the only non-native (*C. familiars*) and one carnivore species (*C. thous*; Figure 2H-I). The other carnivores (*P. cancrivorus* and *N. nasua*) and the hunted species (*D. aurita* and *D. novemcinctus*) had similar capture rates in forests and *cabrucas*, or did not match the expected pattern of distribution (*E. barbara*; Figure 2C-G).

Assemblage metrics in forests and cabrucas

The two-dimensional MDS shows that large mammal assemblages in forests and *cabrucas* were significantly distinct in structure (ANOSIM: $R=0.23$, $p<0.01$; Figure 3A), but not in species composition (ANOSIM: $R=-0.03$, $p=0.66$; Figure S1). The SIMPER analysis revealed that five species (the same for which capture rates varied significantly between the two systems) were responsible for 61% of the variation in assemblage structure between forests and *cabrucas* (Table S1). Capture rates of these species were plotted on the two-dimensional MDS plot (Figure 3B-F).

Contrary to expectations, species turn-over (beta diversity) within *cabrucas* was not significantly greater than within forests (permutation test, *p*=0.67). However, as expected alpha diversity was similar between the two systems (permutation test, *p*=0.44), with an average of 7.6 (± 1.8 SD) species in forest and 7.2 (± 3.6 SD) species in *cabruca* sites (Figure 4A). Thus gamma diversity was also similar, and species accumulation curves were approaching their asymptote after nine samples (Figure 4B).

2.4. Discussion

We recorded 23 species during the study (including the four species fortuitously detected) compared to the 32 species of large mammals that Moura (2003) reported to be present in the largest forest remnants (the current protected areas) in the region. A simplistic interpretation of these numbers indicates that this agroforestry mosaic provides habitat for more than 70 percent of the native large mammal species. However, here we opted to critically assess the conservation value of *cabrucas*, first highlighting the results that help demonstrate the conservation value of these agroforests, second examining why this value should be viewed with caution, and finally discussing the importance of *cabrucas* at a regional scale.

Evidence supporting the conservation value of cabrucas

Our results on assemblage structure, composition and richness suggest that *cabrucas* have a high conservation value: they held as many species as forests in the mixed agroforestry mosaic, there was no significant change in species composition, and no evidence that different management practices led to a higher species turn-over. Although we did observe a significant difference in assemblage structure, these changes were generally much weaker than those previously reported in comparisons of large mammals between mature forest and either secondary forest or diverse shaded agroforests, where both assemblage structure and composition were significantly altered (Harvey *et al.* 2006, Barlow *et al.* 2007, Parry *et al.* 2007). Furthermore, the conservation value of *cabrucas* for large mammals was much higher than alternative anthropogenic land-uses such as urban areas (Crooks 2002), pastures (Daily *et al.* 2003), annual monocultures (Gehring & Swihart 2003), banana (Harvey *et al.* 2006) and eucalyptus plantations (Barlow *et al.* 2007), all of which differ greatly from forests in terms of assemblage structure, composition and species

richness. When compared to such land-uses, *cabrucas* are likely to increase both habitat availability and forest connectivity, increasing the population viability of forest-dependent species.

The records of species of conservation concern also support the high conservation value of *cabrucas*. The widespread presence of the threatened golden-headed lion-tamarin (*Leonthopithecus chrysomelas*) and the endemic marmoset (*Callithix kuhlii*), recorded in all forest and most of *cabrucas* sites, indicates that the agroforestry mosaic harbors a highly interconnected (meta)population of these species. Sparse records of other threatened mammals such as the maned sloth (*Bradypus torquatus*), the yellow-breasted capuchin monkey (*Cebus xanthosternos*) and the thin-spined porcupine (*Chaetomys subspinosus*) suggest that in some situations they might be able to enter and eventually inhabit *cabrucas*. This has been previously reported for *B. torquatus*, in a highly shaded *cabruca* in a farm bordering the Una Biological Reserve (Cassano *et al.* 2011), but the extent to which this species is able to utilize agroforests remains poorly understood.

Hunting pressure could reduce the value of *cabrucas* for large mammals, but we did not find any evidence that the impact of hunting was significantly higher in agroforests than in the forest remnants from the agroforestry mosaic. The two game species analyzed (the opossum, *Didelphis aurita*, and the nine-banded armadillo, *Dasypus novemcinctus*) had similar capture rates in forest and *cabruca* sites, which suggests they are adapted to changes in habitat and that hunting pressure does not decrease their use of the agroforests. The former is not surprising, as both species are common in forest fragments and agroforestry systems, and are well adapted to human-modified landscapes even when open environments are created (Estrada *et al.* 1994, Crooks 2002, Harvey *et al.* 2006, Michalski & Peres 2007, Espartosa 2009). The latter indicates that hunting is not focused on the *cabrucas* where humans are more active, but it might be widespread across land-uses (see further discussion below).

Questioning the conservation value of cabrucas

A number of factors indicate we should be cautious when interpreting our results regarding the high conservation value of *cabrucas*. The high proportion of forest in the study landscape is likely to improve local species richness and abundance. Furthermore, the paired nature of the study could complicate the extrapolation of our results to all *cabrucas* and forest remnants. Especially for highly vagile species, like large mammals, it is likely that the observed differences between environments would be stronger if we had compared the interior of large forest remnants with *cabrucas* isolated from forest patches.

Some species may still require forests, even if observed in *cabrucas*. For example, the capture rate of arboreal species (*C. kuhlii* e *L. chrysomelas*) was lower in *cabrucas* than in forests, indicating *cabrucas* represent sub-optimal habitat. The use of *cabrucas* by both species had been previously recorded (Raboy *et al.* 2004, Raboy *et al.* 2008), including groups of *L. chrysomelas* that had home ranges entirely within *cabrucas* in a landscape comprised mostly by cacao plantations (Oliveira *et al.* 2011). The two species are apparently well adapted to *cabrucas* with a high density of canopy trees (Raboy *et al.* 2004) or a high abundance of key food resources (e.g. jackfruit and bromeliads - Oliveira *et al.* 2011). As tree density and plant species diversity varies greatly between *cabrucas* (Sambuichi 2002, Sambuichi & Haridasan 2007), not all cacao plantations should be expected to provide habitat for these primates. The tayra (*E. barbara*) was also recorded less frequently in *cabrucas*. The species was recorded in cacao and banana agroforests from Talamanca, Costa Rica (Harvey *et al.* 2006), but not in agroforests from southern Mexico (Estrada *et al.* 1994), where it was restricted to the larger forest remnants. The carnivore habit of species not persecuted by humans was not a good predictor of a high ability to use *cabrucas* as we predicted. Further investigation on the vertical use of the vegetation, of prey items and their availability in *cabrucas* should help to disentangle differences in carnivore responses.

Some of the species that benefit from agroforests are of little or no conservation concern. For example, both the non-native domestic dog (*Canis familiaris*) and the crab-eating fox (*Cerdocyon thous*), which is well adapted to open areas (Michalski & Peres 2005, Michalski *et al.* 2006b), had higher capture rates in *cabrucas*. Clearly it is not intuitive to manage forested environments for open-area species such as *C. thous*, while the presence of domestic dogs can have many negative consequences for native fauna, from predation,

competition and the spread of disease (Fiorello *et al.* 2006, Oliveira *et al.* 2008, Vanak & Gompper 2009). Domestic dogs have been widely recorded in forest remnants (Fiorello *et al.* 2006, Whiteman *et al.* 2007, Srbek-Araujo & Chiarello 2008, Espartosa 2009), and their distribution in forests has been related to the abundance they occur in nearby houses (Espartosa 2009) and also to the use of dogs by hunters (Fiorello *et al.* 2006, Whiteman *et al.* 2007). Domestic dogs were present at all properties within the study, and were frequently taken into *cabrucas* by the farm laborers (C.R. Cassano, pers. obs.). The widespread distribution of dogs through both forests and *cabrucas* certainly decrease the conservation value of the agroforestry mosaic for native fauna.

As in many tropical studies, our forest "controls" are likely to have a relatively low conservation value compared to the "original" state of southern Bahia Atlantic forest. Although we recorded a relatively large number of species in the *cabrucas* and the agroforestry mosaic, a significant number of species commonly reported in camera-trap studies from mature forests (e.g.: Silveira *et al.* 2003, Srbek-Araujo & Chiarello 2005, Tobler *et al.* 2008a) were rare or absent in our data. Most of these species are frequently hunted or persecuted, including the ungulates (*Pecary tajacu, Mazama* spp.), large bodied primates and rodents (*Cebus xanthosternos, Cuniculus paca, Dasyprocta* aff. *leporina*.) and felids (*Leopardus* spp., *Puma yagouaroundi*, *Puma concolor*). Other species from these hunted groups that could occur in the area, such as the jaguar (*Panthera onca*), primates (*Alouatta guariba, Brachyteles hypoxanthus*) and ungulates (*Tayassu pecari* and *Tapirus terrestris*; Emmons & Feer 1999), have not been recorded recently even within the largest forest remnants (Moura 2003). The rarity of these large bodied hunted species is further demonstrated by hunter preferences in the study region, as some of the smaller species (e.g. *D. aurita*) are listed amongst the most hunted (Santos 1999). The decline in these large-bodied animals is most likely explained by the local depletion of game stocks (Jerozolimski & Peres 2003), and has also been reported in other landscapes with a high proportion of agroforestry systems (Harvey *et al.* 2006, Bali *et al.* 2007). This "shifted baseline" driven by hunting means we may overestimate the relative conservation value of the *cabrucas*, as these large-bodied species may also be vulnerable to land-use change. Recognition of such shifting baseline is important both to interpret observed results and to set conservation goals (Gardner *et al.* 2009).

Finally, it is important to balance the current conservation value against uncertainty regarding the long term future of *cabrucas* and the mosaic in which they lie. The absence

of hunted mammals decreases large seed dispersal, which is likely to have cascading effects on tree community dynamics in tropical forests (Stoner *et al.* 2007, Terborgh *et al.* 2008). Tree impoverishment has already been observed in small forest remnants in the eastern part of the study landscape (Pardini *et al.* 2009) and elsewhere in the Atlantic forest (Santos 2008). Furthermore, tree regeneration is strongly influenced by management in agroforests, and in older *cabrucas* the composition of the shade trees is increasingly dominated by fast growing native species (pioneers and species found in young secondary forests) and exotic tree species (Sambuichi & Haridasan 2007). Other long-term trends include the thinning of the canopy layer and the replacement of agroforests by other land-uses (Clough *et al.* 2009). This latter activity can result in further deforestation of native forest for the establishment of new cacao plantations. This cacao "boom and bust" cycle (Clough *et al.* 2009) has not yet become widespread in the southern Bahia cacao growing region, despite the long time since the first cacao plantations started in this region. However, it has been occurring on a small scale (either through management practices intensification or land-use change) and remains as a potential threat (Schroth *et al.* 2011).

Implications for the conservation of large mammals in southern Bahia

While our results are intended to provide information about the value of cacao agroforests for large mammals, they are also informative about the importance of these systems in the specific landscape we worked in. The forest-*cabruca* mosaic dominates the landscape between Una Biological Reserve and Serra das Lontras National Park, and potentially constitutes an important biodiversity corridor, increasing habitat availability and decreasing patch isolation for large mammals. However, the scarcity of game species in our study indicates that the functionality of this corridor is severely impaired by past and present hunting pressure and the presence of domestic dogs in the *cabrucas* and forest remnants.

Our results also help underline the importance of maintaining large forest remnants. The large areas of forest may provide the only habitat for species that are sensitive to the synergetic effects of forest fragmentation, overhunting and the invasion of non-native species, while smaller forest remnants might represent higher quality habitat patches within the agroforestry mosaic for some species of conservation concern, such as *L. chrysomelas*. The importance of maintaining forest remnants in this landscape is even higher considering

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that *cabrucas* are agricultural environments, and economic pressures may lead to land-use intensification and change (Schroth *et al.* 2011).

2.5. Conclusion

Given the increasing demand for food and other agricultural products, the implementation of large scale "wildlife-friendly" but low productive agroforestry systems has been questioned (Balmford *et al.* 2005). In this paper we demonstrate how cacao agroforests in a mixed forest and agroforestry landscape can be beneficial for large mammal conservation, considering a range of responses to the conversion of forests to *cabrucas*. Because biodiversity conservation in protected areas will increase if functional connectivity at a broad scale exists, wildlife-friendly land-uses may benefit even species that are unable to use it as permanent habitat, and therefore deserve attention of researches and resource managers. Moreover, diverse and highly shaded agroforests can confer higher resilience to ecosystems and help maintain environmental services (Tscharntke *et al.* 2005). Agroforests also deliver socio-economic benefits to the rural poor being a lower intake and more resilient productive systems (Perfecto & Vandermeer 2008), and providing income from exploitation of other (not the main crop) food and non-food resources (Tscharntke *et al.* 2011).

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Table 1: Large mammals potentially occurring in the forest-agroforest mosaic (following Moura 2003), showing: (1) classification of commonly hunted species, locomotor and diet habits; (2) the number of capture records per type of camera-trap station (on the ground or understory) and per system (forests and *cabrucas*), as well as the number of sites species were recorded; and (3) the predicted and observed responses to conversion of forest to *cabrucas*: arrows indicate greater or smaller capture rates in *cabrucas* than in forests, zero indicates no directional response and "na" stands for species not analyzed.

Species	Hunted	Locomotor [a]	Diet [b]	Understory	Ground	Total	N° sites	Understory	Ground	Total	N° sites	Predicted	Observed
		1. Classification		**2. Forest**				*Cabruca*				**3. Response**	
Order Didelphimorphia													
Didelphis aurita	x	SC	CR	6	30	36	7	6	20	26	7	↓	0
Order Cingulata													
Cabassous sp.	x	SF		0	4	4	2	0	2	2	2	↓	na
Dasypus novemcinctus	x	TE		0	6	6	4	0	2	2	2	↓	0
Euphractus sexcinctus	x	SF		0	0	0	0	0	2	2	1	↓	na
Order Pilosa													
Bradypus variegatus		AR										↓	na
Bradypus torquatus		AR										↓	na
Tamandua tetradactyla		SC											na
Order Primates													
Callicebus melanochir	x	AR	CR									↓	na
Callithrix kuhlii	x	AR	CR	377	83	460	9	161	39	200	8	↓	↓
Cebus xanthosternos		AR	CR	2	1	3	1	3	0	3	1	↓	na
Leontopithecus chrysomelas		AR	CR	290	89	379	9	100	20	120	7	↓	↓
Order Lagomorpha													
Sylvilagus brasiliensis		TE		0	0	0	0	0	1	1	1		na

Table 1 (cont.)

Species	Hunted	Locomotor [a]	Diet [b]	Forest Understory	Forest Ground	Forest Total	Forest N° sites	Cabruca Understory	Cabruca Ground	Cabruca Total	Cabruca N° sites	Response Predicted	Response Observed
Order Rodentia													
Chaetomys subspinosus		AR	FR	0	0	0	0	0	1	1	1	↓	na
Cuniculus paca	x	TE	FR	0	10	10	3	0	15	15	1	↓	na
Dasyprocta aff. *leporina*	x	TE	FR	0	1	1	1	0	0	0	0	↓	na
Guerlinguetus ingrami		AR	CR									↓	na
Hydrochoerus hydrochaeris	x	SA										↓	na
Sphigurus insidiosus		AR	FR	0	0	0	0	1	0	1	1	↓	na
Order Carnivora													
Cerdocyon thous		TE	CR	0	17	17	8	0	50	50	7	↑	↑
Eira barbara		SC	CR	1	43	44	6	0	8	8	3	↑	↓
Galictis vittata		TE	CR									↑	na
Lontra longicaudis		SA	CR										
Leopardus pardalis	x	TE	CR									↓	na
Leopardus sp.	x	SC	CR	0	1	1	1	0	0	0	0	↓	na
Nasua nasua		SC	CR	13	18	31	4	2	0	2	2	↑	0
Potus flavus		AR	CR	4	0	4	2	1	0	1	1	↓	na
Procyon cancrivorus		SC	CR	0	13	13	5	2	23	25	6	↑	0
Puma yagouaroundi	x	TE	CR									↓	na
Puma concolor	x	TE	CR									↓	na
Order Artiodactyla													
Mazama sp.[c]	x	TR	FR	0	1	1	1	0	1	1	1	↓	na
Pecari tajacu	x	TR	FR									↓	na
Total native				**693**	**317**	**1010**		**276**	**184**	**460**			

75

Table 1 (cont.)

| Species | 1. Classification | | | 2. Forest | | | | *Cabruca* | | | | 3. Response | |
	Hunted	Locomotor [a]	Diet [b]	Understory	Ground	Total	N° sites	Understory	Ground	Total	N° sites	Predicted	Observed
Order Carnivora													
Canis familiaris				0	21	21	6	0	66	66	9	↑	↑
Felis catus				0	1	1	1	0	3	3	1	↑	na
Order Artiodactyla													
Bos taurus				0	0	0	0	0	7	7	3	↑	na
Total non-native				**0**	**22**	**22**		**0**	**76**	**76**			

[a] Locomotor habits: AR = arboreal, SC = scansorial, AS = semi-aquatic, SF = semi-fossorial and TE = terrestrial, after Fonseca et al. (1996); [b] Diet: FR = frugivore and CR = carnivore, modified from Fonseca et al. (1996); [c] two species are listed in Moura (2003): *M. americana* and *M. gouazoubira.*
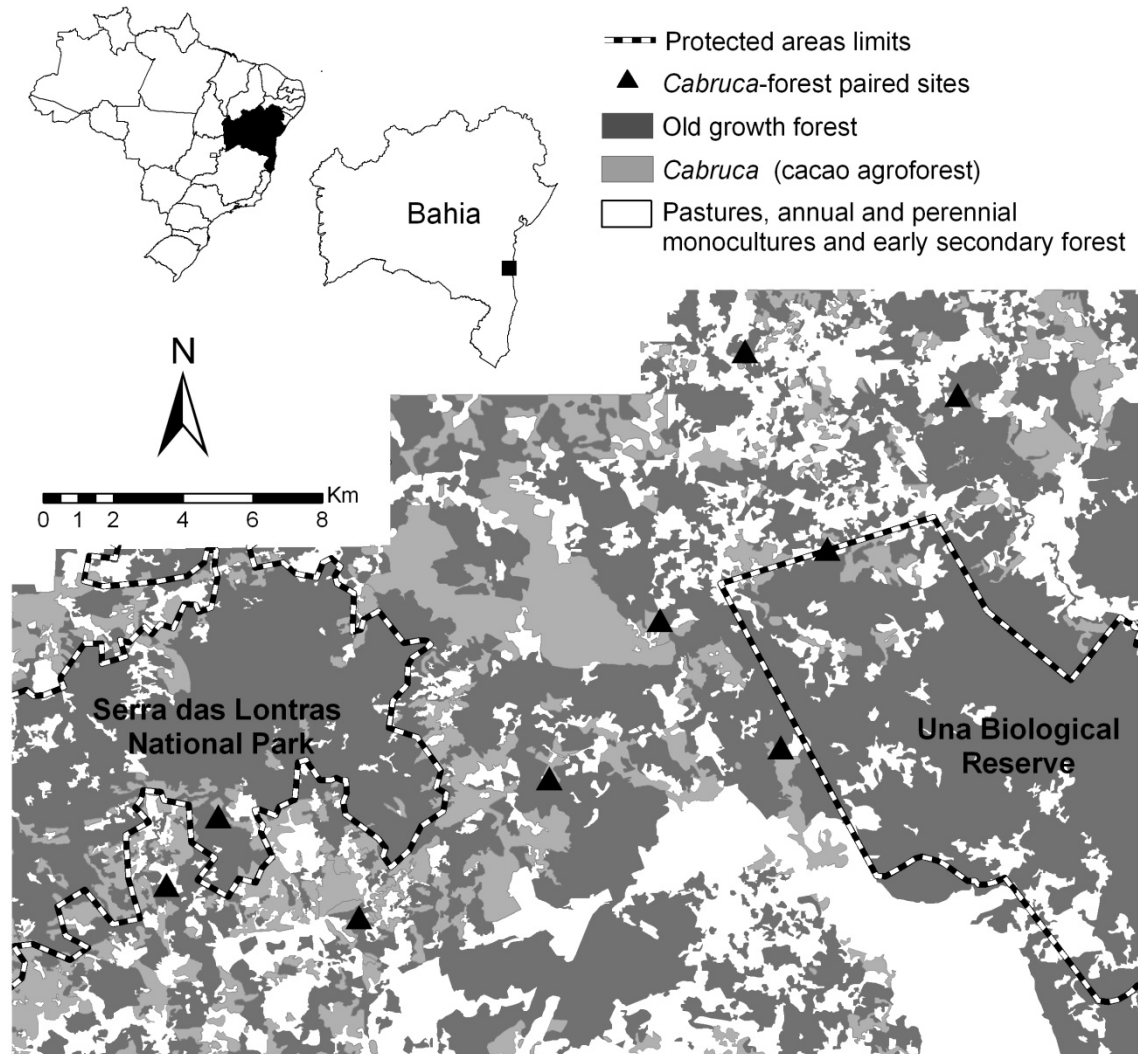
Figure 1: Map of study area, showing the location of the nine *cabruca*-forest sampling blocks in southern Bahia and the main land-uses.
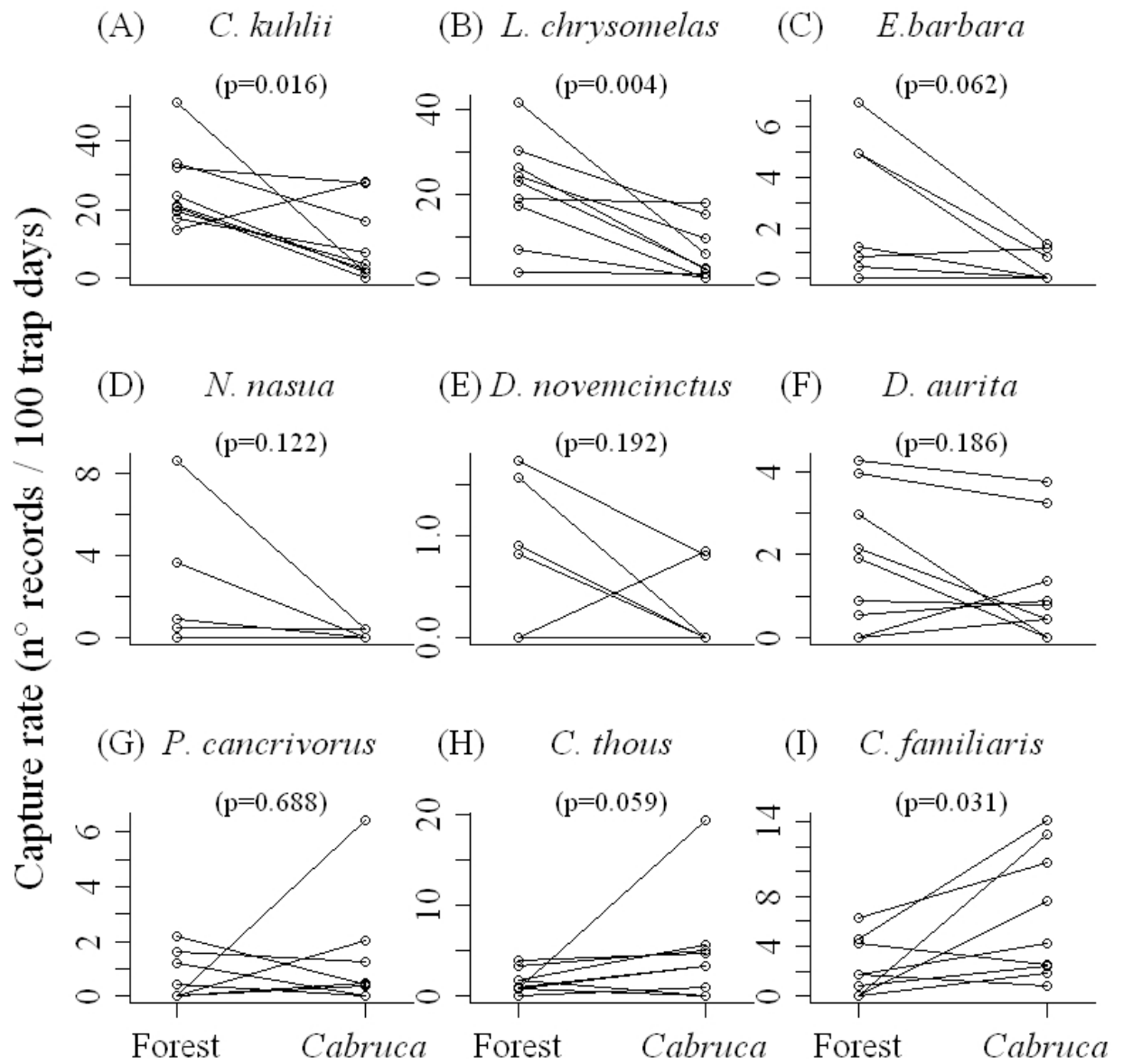
Figure 2: Capture rates of (A-H) eight native and (I) one non-native large mammal species in forest and *cabruca* sites.
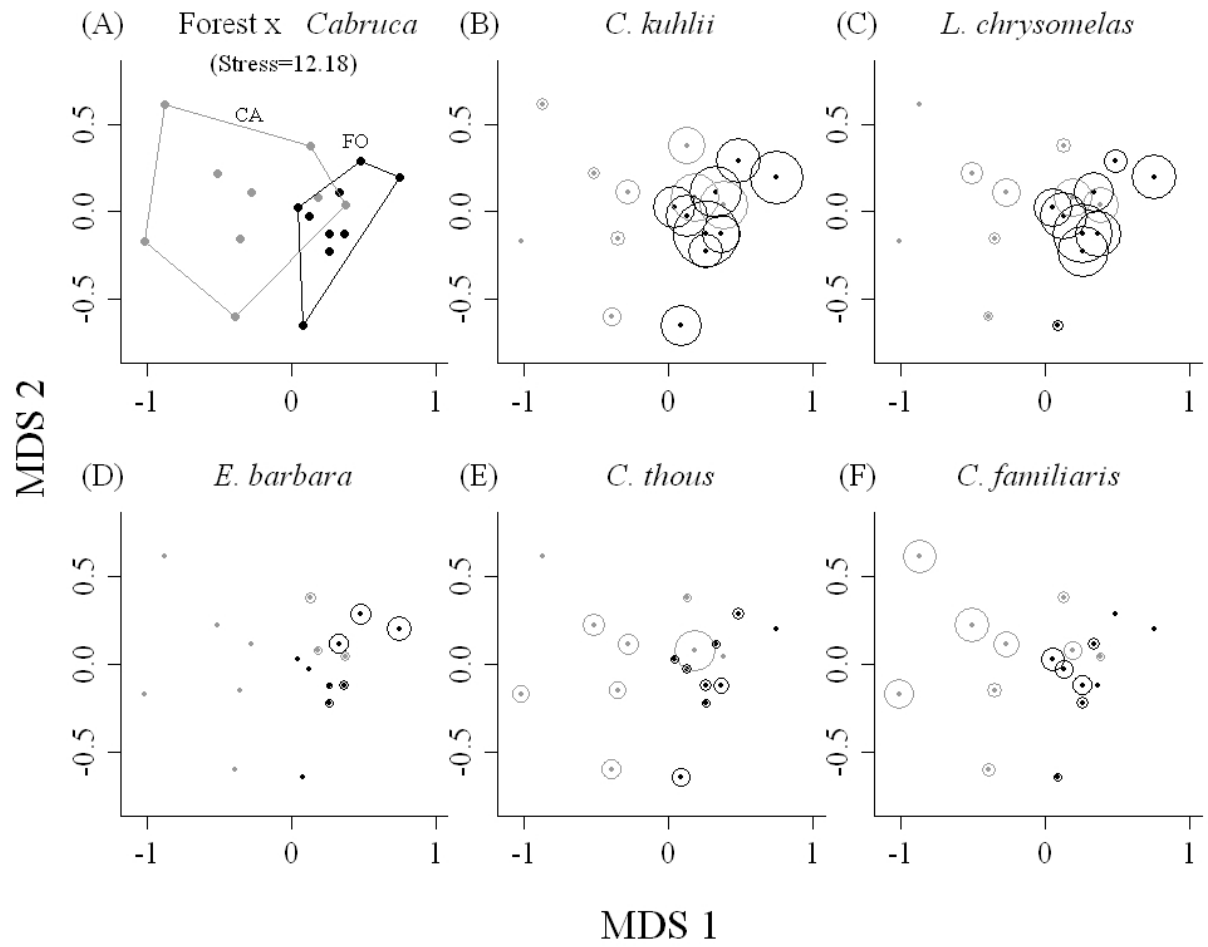
Figure 3: (A) Two-dimensional MDS plots on the similarity in assemblage structure among forest sites (FO) in black and *cabruca* sites (CA) in gray; (B-F) capture rates of species with significant differences between the two systems plotted over two-dimensional MDS (circle diameters are scaled according to square-root of the capture rate).
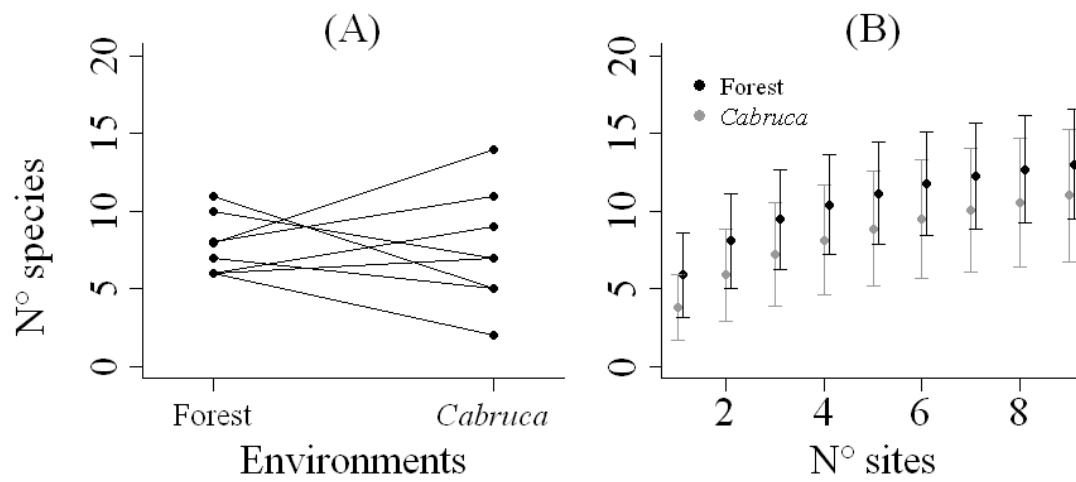
Figure 4: Comparisons of native large mammals richness in forest and *cabruca* sites. (A) Paired comparison of alpha diversity (based on estimated richness by Jackknife1) and (B) Comparison of gamma diversity through species accumulation curves (mean ± 95% CI).

2.6. **Supporting Information**

Table S1: Results from SIMPER analysis, showing the contribution of each species for the dissimilarity in assemblage structure between *cabrucas* and forests.

Species	Species contribution (%)	Cumulated percentage
Leontopithecus chrysomelas	18.42	**18.42**
Callithrix kuhlii	18.2	**36.62**
Canis familiaris	10.18	**46.80**
Cerdocyon thous	7.60	**54.39**
Eira barbara	6.96	**61.35**
Didelphis aurita	5.48	66.83
Cuniculus paca	4.96	71.79
Procyon cancrivorus	4.92	76.71
Nasua nasua	4.78	81.49
Dasypus novemcinctus	3.27	84.76
Bos taurus	3.07	87.83
Cabassous sp.	2.69	90.52
Potus flavus	1.76	92.28
Cebus xanthosternos	1.43	93.71
Felis catus	1.25	94.96
Euphractus sexcinctus	1.13	96.09
Mazama sp.	1.06	97.14
Sylvilagus brasiliensis	0.74	97.89
Dasyprocta aff. *agouti*	0.62	98.5
Leopardus sp.	0.51	99.02
Sphygurus insidiosus	0.50	99.52
Chaetomys subspinosus	0.48	100.00
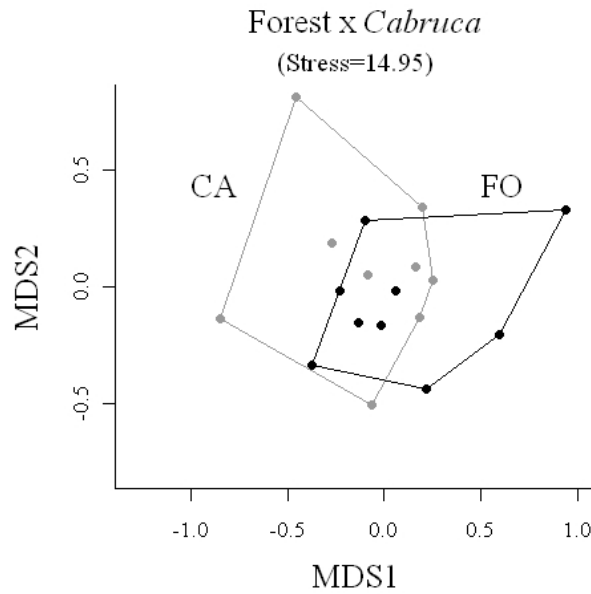
Figure S1: Two-dimensional MDS plots on the similarity in assemblage composition showing a high degree of overlap between forest sites (FO) in black and *cabruca* sites (CA) in gray. The ANOSIM tests did not reveal any significant differences (R=-0.03, p=0.66).

Forest loss or management intensification? Proximal causes of mammal decline in cacao agroforests

Camila Righetto Cassano, Jos Barlow and Renata Pardini

Abstract

The extent and spatial configuration of native vegetation remnants, the intensification of different land-uses and the synergetic effects that result from increasing human use of landscapes exert great influence on biodiversity and associated ecosystem services. Disentangling the influence of deforestation and land-use intensification on species persistence is as important step forward in reconciling conservation and agriculture production. We studied how forest cover and management intensification affect the conservation value of cacao agroforests for mammals in an Atlantic forest landscape in southern Bahia, Brazil. Native mammal assemblages were considerably diverse in agroforests, but hunted and persecuted species were rare. Management intensification had the most important negative effects on large mammal distribution. High domestic dog capture rate, an indirect consequence of management intensification, had negative effect for several species, low canopy connectivity was negative correlated to the distribution of arboreal species and herbaceous vegetation clearance had a minor positive influence on species of non conservation concern. In highly forested regions, restricting the negative effects of land-use intensification, such as the control of domestic dog populations and overhunting, and the careful planning of shade management, is likely to bring greater benefit to native mammals in agroforests than the spatial arrangement of forest remnants. However, the importance of both shade management and remaining forest cover are likely to be higher as deforestation and intensification increase. Current scientific knowledge can already support better policy and management practices to reconcile agricultural production and biodiversity conservation in agroforestry mosaics. Best practices should be continuously adapted from further understanding of agroforestry mosaics and shade management.

Key words: Agroforestry system; domestic dog invasion; landscape management; large mammal; occupancy model; shade management

3.1. Introduction

Land-use change and intensification have led to extensive modification of the Earth's ecosystems and engender great concern regarding the long term conservation of natural resources (Foley *et al.* 2005). An undesirable consequence of such practices is the loss of biodiversity, especially in tropical forest regions (Laurance 2007, Gardner *et al.* 2009). When the original vegetation is converted into alternative land-uses, patches of native vegetation are created and may become functionally isolated, negatively affecting species persistence (Fahrig 2003). While the amount and spatial configuration of the remaining native vegetation is important, population dynamics and chances of species survival in human–modified landscapes is also affected by the intensification of different land-uses (Ricketts 2001, Holzschuh *et al.* 2007, Fischer *et al.* 2011). Incorporating knowledge on how biodiversity is affected by land-use change and intensification into landscape management can enhance conservation outcomes while also meeting goals for agricultural production and improving the maintenance of crucial ecosystem services (Bennett *et al.* 2006, Kellermann *et al.* 2008, Gardner *et al.* 2009).

For decades studies of human-modified landscapes has been dominated by a binary classification of land as either habitat or inhospitable non-habitat, which is also known as the island-model (Lindenmayer & Fischer 2007, Laurance 2008). This approach has allowed the identification of extremely important patterns in human-modified landscapes, such as species-area relationships (Ferraz *et al.* 2007), landscape habitat cover thresholds (Fahrig 2003) and interactions between habitat cover at the landscape and local (patch) scales (Andren 1994, Pardini *et al.* 2010). However, recognition that antropogenic environments can provide habitat for native species and influence community composition and dynamics in forest fragments (Gascon *et al.* 1999, Kupfer *et al.* 2006, Umetsu *et al.* 2008) has led to a growing interest in these environments (collectively known as the matrix). As a general rule, the matrix supports more diverse biological communities and provide more ecosystem services in extensive land-uses (Schulze *et al.* 2004, Harvey *et al.* 2006, Holzschuh *et al.* 2007) and in more forested and heterogeneous landscapes (Faria *et al.* 2007, Holzschuh *et al.* 2007). However, extensive farming uses more land than intensive farming, and the relative benefits of wildlife-friendly farming versus land-use intensification and land sparing has attracted increasing attention in recent years (Green *et al.* 2005, Fischer *et al.* 2008, Phalan *et al.* 2011).

The concept of wildlife-friendly farming is epitomized by agroforestry systems, where forage or crops are cultivated in association with native or planted trees (Somarriba 1992). These systems have been viewed as an important tool for integrating socio-economic and environmental goals in tropical regions (Izac & Sanchez 2001, Perfecto & Vandermeer 2008). From a biodiversity conservation perspective, large areas of cacao (*Theobroma cacao*) and coffee (*Coffea* spp.) agroforests have replaced the original vegetation, but provide important habitat for native tropical forest species (Perfecto *et al.* 1996, Bhagwat *et al.* 2008, Cassano *et al.* 2009, Pardini *et al.* 2009). However, in many regions the long term conservation value of these shaded plantations is threatened by two processes: management intensification, and the loss of native vegetation from land-use change (Clough *et al.* 2009a, Tscharntke *et al.* 2011). The intensification of agroforests often involves the removal of shade trees to attain increasing crop yields, especially in cacao plantations that become less sensitive to sun with age (Tscharntke *et al.* 2011), what can negatively affect native species assemblages (Harvey *et al.* 2006, Bos *et al.* 2007, Harvey & Villalobos 2007, Clough *et al.* 2009b). Alternatively, ongoing land-use change can impoverish native species assemblages in agroforests through decreasing forest cover in the landscape (Faria *et al.* 2007) and increasing isolation from forest remnants (Clough *et al.* 2009b).

While both deforestation and land-use intensification are important processes, often occurring simultaneously, it is important to disentangle their relative influence to effectively guide biodiversity conservation and associated ecosystems services in human-modified mosaics. In their review, Tscharntke *et al.* (2005) highlight that responses to local management practices in agroecosystems are likely to differ from simple to complex landscapes, with implications relevant to agri-environmental schemes (incentives for farmers to benefit the environment). This context-dependency started to receive increasing attention in the last decade, especially regarding the contrast of organic versus conventional agriculture (e.g. Weibull *et al.* 2003, Rundlof & Smith 2006, Holzschuh *et al.* 2007, Fischer *et al.* 2011). Biodiversity responses to management intensification have been the scope of several studies also in cacao and coffee agroforests. However, not only most of these studies treat management intensification as a categorical variable but also very few have simultaneously addressed the influence of native vegetation cover (e.g. Bali *et al.* 2007, Clough *et al.* 2009b).

Here we aim to contribute to the evaluation of the relative importance of forest cover and management intensification for species distribution in agroforestry mosaics. For that we focus on southern Bahia, a region that harbors the most extensive area of diverse shaded cacao plantations in Brazil, locally known as *cabrucas*, with management practices that vary considerably. We examined large mammals as this group is responsible for important ecological processes in tropical forest systems (Terborgh *et al.* 2001, Stoner *et al.* 2007, Nichols *et al.* 2009), and include many "umbrella species" that often form the basis of conservation policies (Simberloff 1998, Roberge & Angelstam 2004, Barua 2011). Large mammals are also vulnerable to changes in land-use (Estrada *et al.* 1994, Daily *et al.* 2003) and to the synergetic effects that result from increasing human use of landscapes, such as hunting activities (Cullen *et al.* 2000, Morrison *et al.* 2007) and the contact with domestic dog populations (Fiorello *et al.* 2006, Vanak & Gompper 2009). We used camera-traps to record large mammals in 30 *cabrucas*, where we measured independent variables describing: (1) the amount and proximity to forest remnants; and (2) direct and indirect aspects of management intensification, represented by vegetation structure and domestic dog capture rate, respectively. We used a model selection approach (Burnham & Anderson 2002) to evaluate alternative hypotheses regarding the relative importance of forest cover and management intensification. In addition, we examine the importance of considering camera-trap detectability by comparing selected sets of occupancy and abundance models (Royle & Nichols 2003, Mackenzie *et al.* 2006), which consider imperfect detectability, to selected sets of models that do not account for this parameter.

3.2. Materials and methods

Study area and sites

Southern Bahia encompasses some of the largest remnants of Atlantic forest in northeastern Brazil, which are recognized for high species richness and endemism (Silva & Casteleti 2003, Martini *et al.* 2007). Part of southern Bahia was once one of the main cacao growing regions in the world, with approximately 6000 km^2 covered by cacao plantations, mainly grown in agroforests. In the late 1900's, Bahian cacao yield dropped as consequence of increasing losses due to witches broom fungal disease (*Moniliophthora perniciosa*) and the low investment that followed the decrease in the international price of cacao (Johns 1999). Since then, a slow process of conversion of the traditional *cabrucas* towards non-shaded or single-species shaded plantation has taken place (Johns 1999).

Nevertheless, landscapes in the region still harbor diverse and dense shaded *cabrucas* mixed with different proportions of native forest (Figure 1A). Two main shade management categories of cacao plantations in the study region are the diverse *cabrucas* and single species (usually *Hevea brasiliensis* or *Erythrina* spp.) shaded plantations. The *cabrucas* are highly heterogeneous, resulting from the shade trees that were kept during their creation and the subsequent tree mortality and recruitment (Sambuichi & Haridasan 2007). Our interest was to evaluate management intensification within these diverse shaded systems, rather than compare diverse agroforests with those shaded by monocultures.

This study was developed in a highly forested landscape (50% forest cover, 15% agroforest; between 39°5'- 39°22' W and 15°4'- 15°14' S; Figure 1B) in the vicinity of two protected areas: the Una Biological Reserve and the Serra das Lontras National Park (for further information on landscape features, original vegetation and climate see Cassano *et al.*, unpublished). Sample sites were located in 30 *cabrucas* with different management practices and surrounding landscape contexts. Two initial criteria for site selection were: (1) cacao plantations mainly shaded by native tree species (i.e. considered a *cabruca*); and (2) minimal distance of 1.5 km between sites. We maximized the variability and minimized spatial correlation of surrounding contexts by dividing the landscape into three groups (west, central and east) and searching for sites distant to forest and near small and large forest remnants within each group.

Data collection

Vegetation samples

Vegetation was sampled in four 10 x 25 m plots at each site to quantify direct consequences of management intensification. The plots were spaced every 20 m in an area of 25 x 100 m where two trap stations were placed (see section Mammal sampling). Within each plot, we counted and measured all shade trees and banana stems ≥ 10 cm diameter at breast height (DBH). We identified the most common exotic trees and classified them according to whether they produce fleshy (*Annona squamosa*, *Artocarpus heterophyllus*, *Carica* sp., *Citrus* spp., *Spondias mombin*) or non-fleshy fruits (*Erithryna* spp., *Gliricidia* sp. and *Hevea brasiliensis*). The remaining native trees were not identified. We also counted all cacao shrubs inside half of each plot (5 x 25 m). We counted the number of medium and large bromeliads on tree crowns and estimated the proportion of each tree

crown connected to adjacent trees using an index ranging from 0 to 4, corresponding to the number of quadrants of the tree crown connected to adjacent trees. The same index was used to estimate connectivity among cacao shrubs. We measured the height of herbaceous vegetation, the tallest herbaceous plant within a 0.5 x 0.5 m quadrate, at 12 points that were evenly distributed at 5 m intervals along one of the plot sides and along its center. We interviewed farm laborers for an estimation of the frequency of herbaceous vegetation clearance. Shade trees, banana stems and cacao trees were measured just once, at the end of the second sampling session (see section Mammal sampling). Herbaceous vegetation height, highly variable in time, was measured at the end of the second, third and fourth sampling sessions.

Mammal sampling

Two camera-traps (analog Trapacamera - www.trapacamera.com) were placed in each sampling site, one on the ground and one in the understory, following the sampling protocol described in Cassano *et al.* (unpublished). Data on native and non-native mammals were collected on four sampling sessions, during July-October of 2007 and 2008, and January-April of 2008 and 2009. For logistical reasons, sampling was divided into three groups of sites, each composed of 10 *cabrucas* that were sampled simultaneously. During each sampling session the trap stations of all 30 *cabruca* sites were active for four weeks and checked weekly to exchange films and re-bait. The malfunction of camera-traps and the depletion of films before the weekly check prevented equal effort among sites, sessions and stations, which was taken into account in data analysis.

Data analysis

Forest cover

We used two forest cover variables to quantify the amount and proximity of forest remnants around each *cabruca* site: (1) forest cover index – quantifies the area covered by native forest in the immediate vicinity of *cabruca* sites (400-m radius circumferences), weighted by the distance to the sites (Table 1); and (2) proximity to large forest remnants – a categorical variable distinguishing, at a larger spatial scale, *cabruca* sites near (≤800 m) or far (>800 m) from the main forest remnants in the landscape (Figure 1B; see supporting information (SI) for details on forest cover index and proximity to large forest remnants

calculations). Data were extracted from a digital land-use map provided by the Instituto de Estudos Socioambientais do Sul da Bahia, based on aerial photographs from years 1996/97 and Ikonos satellite image from 2007, using ArcGIS 9.2 (ESRI 2007).

Management intensification

In order to select appropriate metrics of management intensification, we used a principal component analysis (PCA) to ordinate sites according to the 14 variables describing vegetation structure and composition (Table 1). From this analysis, we chose two vegetation measurements that are both impacted by management intensification and have ecological meaning to animals: (1) sum of canopy connectivity index (directly related to shade management); and (2) average herbaceous vegetation height (directly related to frequency of herbaceous vegetation clearance). Increasing canopy connectivity is expected to positively influence the ability of species that move through the vegetation (arboreal and scansorial). Increasing herbaceous vegetation height is expected to influence species that move on the ground (terrestrial and scansorial) either positively, providing shelter, or negatively, decreasing prey availability.

Domestic dog capture rate, calculated as the total number of records by total sampling effort at ground trap stations (Table 1), was also inserted into the PCA, providing inference about the relationship between this variable and management intensification. We considered one record the presence of at least one picture during a 24-hour period (starting at 00:00 h ending at 24:00 h) and sampling effort was computed (in trap-days) counting only days in which there was no apparent camera-trap malfunction or film depletion. PCA was run in a correlation matrix using Package Vegan and R 2.10.1 (R Development Core Team 2009).

Modeling large mammal distribution as a function of forest cover and management intensification

We used abundance models (Royle & Nichols 2003) to investigate the influence of the five explanatory variables (two forest cover and three management variables) on species abundance (λ), and single-season occupancy models (Mackenzie *et al.* 2006) to investigate their influence on species occupancy (Ψ) and detectability (p). Abundance models were used for one species recorded in 29 sites, while occupancy models were used for species recorded in more than five and less than 26 sites (Table S1). The detectability from occupancy models was modeled as a function of explanatory variables only for species recorded in more than 10 sites. This parameter was modeled as a function of explanatory variables because both forest cover context and management intensification are expected to change the use of the agroforests by mammals, which can influence species detectability. Additionally, the detectability from abundance and occupancy models were modeled as a function of sampling effort, which varied between sites and sessions. We controlled for uneven sampling effort in ground and understory stations by excluding extra days from the station with the larger sampling effort before accounting for detection or non-detection of the target species in each of the four sampling sessions. We assumed that (1) changes in occupancy and abundance during the period of the study are irrelevant for the analysis owing to the slow population dynamics and the long life expectancy of large mammals; and (2) that the study design guarantees the independence of the probability of detecting a species among sites. Given the small size of the sampling sites in relation to the home range of most large mammals, we interpreted occupancy estimates as the proportion of sites used by the species (for further discussion see Mackenzie *et al.* 2006, p.105), assuming random movement of the target species in and out of a study site during each sampling session. The parameters λ from abundance model and p from occupancy models are interpreted as a proxy of site use intensity.

We also modeled species naïve occurrence (presence/absence) and the number of records (number of trap-days the species was recorded, considering only the ground trap station for terrestrial species and both ground and understory stations for arboreal and scansorial species) using general linear models (GLMs) and mixed models (GLMMs) (Table S1; see SI for detail on GLMs and GLMMs) to investigate how detectability affects the results. Hereafter we refer to these models as naïve occurrence and record count models, and we

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use the general term distribution when referring to both where a species occurs and the intensity it uses the sampling sites.

The dependant variables λ, Ψ, p, naïve occurrence and number of records were modeled using single and multi-variable models, and one constant reference model. Multi-variable models included additive combinations of two or three explanatory variables, but never included the two forest cover variables together as these quantify the same factor. We avoided multi-variable models with higher numbers of variables or interaction terms given the limited number of sampling units. The set of candidate models differed for arboreal, scansorial and terrestrial species, because we did not include canopy connectivity in models for terrestrial species or herbaceous vegetation in models for arboreal species (Table S1).

We compared alternative models using the difference in their AICc (Akaike's Information Criterion modified for small samples; Burnham & Anderson 2002) values in relation to the best model (ΔAICc). A value of ΔAICc ≤ 2 indicates equally plausible models. We standardized all explanatory variables dividing by maximum value before running the analysis. Models that did not converge using the default optimization method, Broyden–Fletcher–Goldfarb–Shanno (BFGS), were optimized by the method simulated annealing (SANN; Bélisle 1992). We selected the plausible occupancy models in a three-step procedure (see SI for details). R 2.10.1 (R Development Core Team 2009) was used to run the occupancy and the abundance models (using the Unmarked package), as well as the GLMs and GLMMs.

3.3. Results

The final effort per site ranged from 168 to 254 trap-days, with a total sampling effort of 6592 trap-days. We obtained a total of 1694 records from 20 native and 267 records from four non-native mammals. Wied's black-tufted-ear marmoset (*Callithrix kuhlii*), golden-headed lion-tamarin (*Leontopithecus chrysomelas*), crab-eating fox (*Cerdocyon thous*) and opossum (*Didelphis aurita*) were the most common native species, representing 89% of the records from this group (Table 2). Domestic dog (*Canis familiaris*) was the most frequently recorded non-native species, representing alone 88% of the records from this group. It was also the third most common species overall, and the only species recorded at all sites (Table 2).

Cabruca *management*

The first axis of the PCA explained 28% of the variation in vegetation structure and domestic dog capture rate among sites. This axis represents a gradient of management intensity, with increasing values representing the replacement of canopy trees by banana stems in the shade layer, and the increase in number of cacao shrubs and in the frequency of herbaceous vegetation clearance. The canopy connectivity and the herbaceous vegetation height were both negatively correlated with this axis, while domestic dog capture rate was positively correlated (Figure S1). The second axis of the PCA explained 16% of the variation in vegetation structure and also correlates with plantation management. Values for this axis increase from *cabrucas* shaded by a larger number (and size) of native trees to those shaded by a larger number of exotic non-fruit trees. Canopy connectivity was negatively correlated and herbaceous vegetation height positively correlated with this axis.

Large mammal distribution as a function of forest cover and management intensification

Nine species were recorded in six or more sites and were included in the analysis. We modeled abundance and number of records for one species: *C. kuhlii*; occupancy, detection probability, naïve occurrence and number of records for five species: the previously mentioned *C. thous*, *D. aurita* and *L. chrysomelas*, plus tayra (*Eira barbara*) and crab-eating raccoon (*Procyon cancrivorus*); and only occupancy and naïve occurrence for three species: naked-tailed armadillo (*Cabassous* sp.), paca (*Cuniculus paca*) and wild cat (*Leopardus* sp.).

Arboreal species (*C. kuhlii* and *L. chrysomelas*) responded positively to the increase in the canopy connectivity index and the decrease in domestic dog capture rate (Table 3), variables that are directly and indirectly related to management intensification, respectively. For *L. chrysomelas*, decrease in domestic dog capture rate had a positive effect on both occupancy and naïve occurrence. Detectability of *L. chrysomelas* was positively related to canopy connectivity, while number of records was positively related to both the increase in canopy connectivity and the decrease in domestic dog capture rate. Abundance models selected for *C. kuhlii* always contained the canopy connectivity index, either alone or together with domestic dog capture rate. For this species, the record count

model including the domestic dog capture rate was the best model, although it was equally likely as the reference model (Table 3).

For scansorial species (*D. aurita*, *E. barbara*, *Leopardus* sp. and *Procyon cancrivorus*), the reference model was always part of the selected sets of occupancy, naïve occurrence and record count models, with the exception of naïve occurrence of *Leopardus* sp. (Tables 3 and S2). Naïve occurrence of this species was negatively affected by management intensification, being negatively correlated to the increase in domestic dog capture rate and decrease in herbaceous vegetation height, as well as by forest cover context, being negatively affected by the proximity to large forest remnants (Table 3). For this species, the occupancy model with the lower AICc also had occupancy negatively affected by domestic dog capture rate and an evidence weight twice as large as the reference model. Moreover, herbaceous vegetation height and proximity to large forest remnants also appeared in plausible models with the same effects described to naïve occurrence. For *D.aurita*, the reference model had the lower AICc, independently of the parameter modeled. The selected sets of occupancy, naïve occurrence and record count models for *E. barbara* and *P. cancrivorus* were composed by a large number of plausible models (3-8) and few occupancy models for *P. cancrivorus* converged when the default optimization method was used. When using the SANN optimization method, different sets of models were selected every time the models were rerun and a new model selection performed. The SANN optimization method was also used in non-convergent occupancy models for *Leopardus* sp., but the results from model selection did not change when models were rerun.

Responses to forest cover and management intensification differed among terrestrial species (*Cabassous* sp., *Cuniculus paca* and *Cerdocyon thous*). For *Cabassous* sp. occupancy was negatively affected by the increase in herbaceous vegetation height and domestic dog capture rate, showing opposite responses to a direct and an indirect consequence of management intensification. Occupancy of *Cabassous* sp. was also negatively affected by proximity to large forest remnants. The same three variables explained *Cabassous* sp. naïve occurrence (Table 3). For *C. paca*, the selected sets of occupancy and naïve occurrence models were composed of seven plausible models each, with different combinations of the same three variables, and the reference model had the lower AICc (Table S2). *C. thous* was negatively affected by the increase in forest cover, with occupancy and detectability negatively correlated to proximity to large forest

remnants and forest cover index, respectively (Table 3). For this species, naïve occurrence was negatively influenced by proximity to large forest remnants and the increase in herbaceous vegetation height, while the number of records was negatively influenced by the increase in forest cover index. However, for both naïve occurrence and number of records the selected set of models included the reference model.

3.4. Discussion

Our results confirm that although *cabrucas* harbor a diverse large mammal assemblages, many important species are absent from this land-use (cf. Cassano *et al.* unpublished). This previous study suggests the scarcity of ungulates, large-bodied primates and rodents, and felids in the agroforestry mosaic as a consequence of overhunting and persecution, and states the need to consider this baseline when analyzing current responses to environmental variables. In the following section we briefly consider the strength and limitations of different classes of models for analyzing camera-trap data. We then discuss the relative importance of forest cover and management intensification for the use of agroforests by large mammals, place the results into the context of a highly forested although defaunated landscape, and highlight the implications of management practices for the long term conservation of native large mammals in agroforestry mosaics.

Comparing models that consider or not detectability

Results from models that consider or and models that do not consider detectability were similar regarding the variables that best explained mammal use of *cabrucas*, the direction of their effects, and the selection of the reference model. Thus, although occupancy estimate can be improved by models that consider imperfect detectability (Bailey *et al.* 2004), general responses seem to be identifiable in the absence of this parameter, at least when sampling is standardized among sites as in this study. Our results also show that detectability retains information about the intensity of use of sampling sites and that occupancy models can be improved when both Ψ and p are modeled as a function of environmental variables. High costs and limited technical and human resources mean that many ecological studies have a limited sampling size, and the complexity of models that consider detectability might prevent model convergence, as in the case of occupancy models for crab-eating raccoon (*P. cancrivorus*). Models based on detectability might also

complicate the use of robust sampling designs (e.g. nested or paired designs), which can also improve the identification of significant biological patterns. While occupancy models are undoubtedly useful for analyzing camera-trap presence/absence data, our data suggests we should not discard information about the intensity of use contained in counts or rates of records, especially when robust and standardized sampling design are used. As with other statistical paradigms, there is a case for plurality, and decisions about which statistics should reflect the characteristics of the data collection protocols and the biological system that is being studied (cf. Stephens *et al.* 2005).

Forest cover vs. management intensification

Management intensification had the most important negative effects on large mammal distribution in *cabrucas*. Domestic dog capture rate, which represents an indirect effect of management intensification, affected the larger number of species while canopy connectivity affected arboreal species. Although domestic dogs are considered weak competitors, they may be important competitors and predators of native species because high population densities are subsidized by humans (Vanak & Gompper 2009). Pathogen spillover from dogs is also a substantial threat to native fauna (Fiorello *et al.* 2006, Whiteman *et al.* 2007, Curi *et al.* 2010). Negative correlations between the distribution of domestic dog and native mammals have been previously detected, and this non-native species has been reported more frequently in forest edges than in interiors, in the matrix than in forest remnants, and near human settlement (Espartosa 2009, Lacerda *et al.* 2009, Vanak & Gompper 2010). However, the negative effects might spread through the landscape if areas where domestic dogs occur function as a sink of individuals or if pathogens spread through native populations.

The negative influence of reduced canopy connectivity for arboreal species is especially relevant given the relatively high number of threatened and endemic species in this group (Fonseca *et al.* 1996, Costa *et al.* 2005). Habitat and inter-patch connectivity provided by agroforests are crucial for long term viability of neotropical primate populations inhabiting the study region and agroforestry landscapes elsewhere (Estrada *et al.* 2006, Zeigler *et al.* 2010). Our study confirms previous results that underline the importance of canopy pathways for arboreal species inhabiting agroforests (Estrada *et al.* 2006). Nevertheless, because the canopy layer in *cabrucas* are mainly composed of native trees, increasing canopy connectivity might also increase food availability, and arboreal species should not

be expected to positively respond to canopy connectivity provided exclusively by exotic non-fleshy fruit trees which are common in cacao agroforests.

Management intensification had also positive effects on mammals. However, the species that positively responded to the decrease in herbaceous vegetation height has no conservation concern. Armadillos have been recorded in highly intensified agroecosystems and modified landscapes (Estrada *et al.* 1994, Silva & Pontes 2008, Andrade-Nunez & Aide 2010), and also occur in open biomes (Fonseca *et al.* 1996). Thus, management intensification of agroforests is likely to lead to a gradual decline in the conservation value of this system, as intensively managed areas are used by fewer species of conservation concern and are more prone to invasion by open-area generalist mammals.

On the other hand, large mammals were not limited by the spatial arrangement of forest remnants in our study. Lower amount of surrounding forest cover and increased distance to main forest remnants improved the use of agroforests especially by generalist species, but did not restrict the distribution of forest or threatened species. Our results differ from previous studies that contrasted the relative importance of forest cover and management intensification to vertebrate distribution among cacao and coffee agroforests. Distance to a main forest remnant was the most important factor and negatively affected richness and abundance of large mammals in coffee agroforests from Western Ghats mountain ranges in India (Bali *et al.* 2007). In Central Sulawesi cacao agroforests, forest specialist birds responded positively to forest edge proximity, but shade management also played important role for most functional groups (Clough *et al.* 2009b). We examine reasons for these differences in the following section.

Placing results into regional context

The responses of individual species and species assemblages to changes in local scale variables may depend on landscape context, which can influence population dynamics and species pool (Andren 1994, Tscharntke *et al.* 2005). For example, higher isolation from nesting habitats in homogeneous landscapes increased the effect of wheat field management on bee species richness (Holzschuh *et al.* 2007), while the abundance and richness of specialist small mammals increased with forest fragment size in one landscape with intermediate forest cover (~30%), but not in highly forested or deforested landscapes (Pardini *et al.* 2010). Given the large home range sizes and the high vagility of mammals,

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even the larger-scale variable proximity to main forest remnants do not segregate *cabruca* sites into broad scale contexts equivalent to the landscape variables used in these studies. Because our study region harbors approximately 65% of tree cover (forests plus agroforests), our results lie in the context of a highly forested landscape, which is likely to reduce species responses to local scale variables, such as management intensification and local-scale forest cover (Tscharntke *et al.* 2005, Pardini *et al.* 2010, Batary *et al.* 2011).

Few studies have evaluated the importance of local management intensification across a gradient of forest cover at broad scales, especially regarding large bodied species (Batary *et al.* 2011). Given the scarcity of studies, the non linearity of biological responses to environmental variation and the fact that different biological groups perceive the environment at distinct scales, generalizations regarding the relative importance of land-use change and intensification and the spatial arrangement of land-mosaics are difficult. This uncertainty complicates the debate about wildlife-friendly versus land-sparing approaches as strategies to reconcile agricultural production and conservation. As previously discussed by other authors (e.g. Green *et al.* 2005, Fischer *et al.* 2008) no single better approach is likely to exist and different balances of the two strategies should be adapted to different contexts.

Besides the amount of remaining forest at broad spatial scales, overhunting may also influence our results if hunted species are also vulnerable to the conversion of forest into *cabrucas*, or to the direct and indirect consequences of agroforest intensification. Moreover, responses to management intensification might also be stronger if wider ecological gradients of management intensity were considered. While we restricted our sampling to the *cabrucas*, we are aware that plantations shaded exclusively by rubber trees and unshaded plantations are increasing in southern Bahia (Johns 1999, Schroth *et al.* 2011).

Improving agroforestry mosaics for large mammals

Our findings indicate that the spatial arrangement of forest remnants plays a minor role in determining the conservation value of *cabrucas* in our study system. Although the maintenance of forest cover is likely to be critically important in highly deforested landscapes, it should not be the sole target of conservation activities in highly forested landscapes. For conservation within agroforests, shade management seems to be a more important practice, as it influences the maintenance of additional habitat and increase inter-patch connectivity for arboreal species. However, the viability of this practice should be analyzed cautiously. Since canopy connectivity was correlated with the number of shade trees, an intuitive way to achieve high canopy connectivity would be the maintenance of high tree densities. Nevertheless, because tree density is negatively correlated to cacao yield (Clough *et al.* 2011), greater benefit from shade management would be obtained from the planning of pathways and a careful selection of tree species when tree pruning or removal is necessary (Cassano *et al.* 2009, Clough *et al.* 2011).

Some management practices can improve the conservation value of agroforests and agroforestry landscapes without altering land-use type and management. Domestic dog vaccination and sterilization campaigns are frequently suggested as effective conservation practices for native mammals (Curi *et al.* 2010, Vanak & Gompper 2010). In agroforestry landscapes, such practices would produce multiple benefits by increasing agroforest use and permeability and reducing edge effects on forest remnants as suggested in previous studies (e.g. Woodroffe & Ginsberg 1998, Lacerda *et al.* 2009). A similar benefit is likely to be gained from the control of hunting activities, which has been suggested as an important pressure preventing the use of agroforestry landscapes by numerous species (Estrada *et al.* 1994, Harvey *et al.* 2006, Cassano *et al.* unpublished). Hunting activities are biased towards large-bodied species, and are known to impact species of low reproductive rate (Robinson & Redford 1986, Jerozolimski & Peres 2003). A mixed approach of law enforcement, awareness of regionally vulnerable species, and developing a supply of alternative sources of fresh meat, could help to reduce overhunting.

3.5. Conclusions

We analyzed the proximal causes that determine the use of agroforests by large mammals, and examine the regional characteristics influencing the observed responses in order to produce more general conclusions. Our results suggest that the control of domestic dog population and hunting activities would bring great benefit to large mammal conservation in agroforestry landscapes without influencing the productivity of agroforests. In highly forested regions, the spatial arrangement of forest remnants has minor effects on mammal populations inhabiting the agroforestry mosaic, and shade improvement in agroforests should be adopted taking into account the trade-off between productivity and biodiversity conservation. Win-win situations in cacao yield and species richness have been recently reported in agroforests from Sulawesi, Indonesia (Clough *et al.* 2011). However, the thinning of the canopy layer, the replacement of agroforests by other land uses, and the establishment of new plantations in current forest lands remain as long term threats in cacao and coffee growing regions (Clough *et al.* 2009a, Schroth *et al.* 2011, Tscharntke *et al.* 2011). In this scenario of future land-use change and intensification, forest cover and shade maintenance in agroforests are likely to gain greater importance. The understanding of emergent properties from agroforestry mosaics and shade management would be improved by additional studies in different landscape and local contexts. Nevertheless, sound basis management can already be drawn from current scientific knowledge, and this information should be incorporated into policy and applied by land managers.

Acknowledgments

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Table 1: Minimum, mean and maximum values of forest cover index, 14 vegetation variables and domestic dog capture rate among the 30 *cabruca* sites. Variables in bold were used in single and multi-variable models to explain large mammal distribution among *cabrucas*.

Variable	Min.	Mean± SD	Max.
Forest cover index	**0**	**668 ± 483**	**2006**
Sum of canopy connectivity index [a]	**1.0**	**27.5 ± 24.7**	**119.0**
Average ground vegetation height (cm)	**22.0**	**69.9 ± 31.5**	**140.0**
Number of native trees[a]	4	10.5±6.9	34
Number of exotic non-fleshy fruit trees [a]	0	2.1±2.9	10
Number of exotic fleshy fruit trees [a]	0	1.4±2.0	8
Sum of basal area of native trees (m^2) [a]	0.2	1.4±0.7	2.7
Sum of basal area of exotic non-fruit trees (m^2) [a]	0	0.2±0.2	1.0
Sum of basal area of exotic fruit trees (m^2) [a]	0	0.2±0.4	1.7
Number of medium size bromeliads on tree crowns [a]	0	4.7±4.7	19
Number of large size bromeliads on tree crowns [a]	0	4.0±5.0	22
Number of cacao shrubs [b]	18	69.9±31.5	140
Sum of cacao connectivity index [b]	24	87.6±28.7	143
Number of banana stems [a]	0	4.0±4.2	15
Frequency of herbaceous vegetation clearance (times/year)	0.5	1.3±0.6	2.5
Domestic dog capture rate (records/100 trap-days)	**0.8**	**7.1 ± 6.0**	**19.1**

[a] Variables measured in 0.1 ha; [b] variables measured in 0.05 ha

Table 2: Native and non-native mammals registered in 30 *cabrucas*. Number of records (in trap-days) in understory and ground trap stations and in total, number of detections (number of sessions with records), and number of sites where each species was recorded. Locomotor habit: AR = arboreal, SC = scansorial, and TE = terrestrial.

Species	Understory	Ground	Total	N° detection	N° sites	Locomotor habit
Order Didelphimorphia						
Didelphis aurita [b]	48	81	129	40	21	SC
Order Cingulata						
Cabassous sp. [a]		17	17	11	7	TE
Dasypus novemcinctus		5	5	5	5	TE
Euphractus sexcinctus		2	2	1	1	TE
Order Pilosa						
Tamandua tetradactyla		2	2	2	1	SC
Order Primates						
Callithrix kuhlii [c]	626	191	817	81	29	AR
Cebus xanthosternos	3		3	2	1	AR
Leontopithecus chrysomelas [b]	289	39	328	52	21	AR
Order Lagomorpha						
Sylvilagus brasiliensis		1	1	1	1	TE
Order Rodentia						
Chaetomys subspinosus		1	1	1	1	AR
Cuniculus paca [a]		27	27	9	6	TE
Sphygurus insidiosus	1		1	1	1	AR
Order Carnivora						
Cerdocyon thous [b]		226	226	54	25	TE
Eira Barbara [b]	18	33	51	17	11	SC
Nasua nasua	6	2	8	5	4	SC
Leopardus sp. [a]	1	8	9	7	6	SC
Potus flavus	1		1	1	1	AR
Procyon cancrivorus [b]	4	59	63	29	19	SC
Order Artiodactyla						
Mazama sp.		2	2	2	2	TE
Pecari tajacu		1	1	1	1	TE
Total native species	997	697	1694			
Order Carnivora						
Canis familiaris		235	235	72	30	
Felis catus		4	4	3	2	
Order Artiodactyla						
Bos taurus		18	18	8	7	
Ordem Perissodactyla						
Equus sp.		10	10	7	6	
Total non-native species		267	267			

[a] Species analyzed for Ψ and naïve occurrence only; [b] species analyzed for Ψ, p, naïve occurrence and capture rate; [c] species analyzed for λ and capture rate.

Table 3: Selected models (ΔAICc ≤ 2) describing the variation in the use of 30 *cabruca* sites by large mammals as a function of forest cover, management intensification, and sampling effort. Only shown results of species if the reference model was not included among the selected models for at least one model class analyzed. For the remaining species see Table S2. Dependant variables: λ = abundance, Ψ = occupancy, p = detection probability, occ = naïve occurrence and rec = number of records. Explanatory variables: fo = forest cover index; prox = proximity to large forest remnants; can = canopy connectivity index; herb = herbaceous vegetation height; dog = domestic dog capture rate; eff = sampling effort. The notation (.) stands for constant.

Species	Model description	Model class	AICc	ΔAICc	w	fo	prox	can	herb	dog	eff
						Forest cover		**Management intensification**			
Callithrix kuhlii	λ(can); p(.)	abundance	144.95	0.00	0.28			1.77			
	λ(can+dog); p(.)		146.90	1.96	0.11			1.40		-0.57	
	rec (dog)	count	306.46	0.00	0.23					-1.57	
	rec (.)		307.35	0.89	0.15						
	rec (fo+dog)		307.39	0.93	0.14	-1.49				-1.45	
	rec (fo)		307.72	1.26	0.12	-1.68					
Leontopithecus chrysomelas	Ψ(dog); p(can)	occupancy	144.33	0.00	0.34			4.09		-2.95	
	Ψ(.); p(can)		145.88	1.56	0.16			4.36			
	occ (dog)	naïve occurence	36.26	0.00	0.28					-2.88	
	rec (dog)	count	186.62	0.00	0.40					-3.43	
	rec (can+dog)		188.39	1.77	0.16			2.03		-2.97	

Table 3 (cont.)

| Species	Model description	Model class	AICc	ΔAICc	w	Forest cover		Management intensification			eff
						fo	prox	can	herb	dog	
Leopardus sp.	Ψ(dog); p(.)	occupancy	52.67	0.00	0.13					-5.85	
	Ψ(prox+dog); p(.)		53.71	1.04	0.08		-1.86			-6.95	
	Ψ(fo+dog); p(.)		53.83	1.16	0.08	-12.56				-16.00	
	Ψ (dog+herb); p(.)		54.26	1.59	0.06				13.58	-9.38	
	Ψ(herb); p(.)		54.27	1.60	0.06				16.91		
	Ψ(.); p(.)		54.45	1.78	0.06						
	occ (dog)	naïve occurence	27.03	0.00	0.15					-5.71	
	occ (prox+dog)		27.90	0.87	0.09		-1.49			-6.46	
	occ (prox+dog+herb)		28.96	1.92	0.06		-2.55		4.38	-5.67	
Cerdocyon thous	Ψ(.); p(fo)	occupancy	162.56	0.00	0.22	-2.04					
	Ψ(prox); p(fo)		163.08	0.52	0.17	-2.33	-5.36				
	occ (.)	naïve occurence	29.18	0.00	0.23						
	occ (prox)		30.50	1.32	0.12		-0.98				
	occ (herb)		30.81	1.63	0.10				-1.76		
	rec (fo)	count	189.15	0.00	0.30	-1.69					
	rec (.)		189.88	0.73	0.21						
Cabassous sp.	Ψ(prox+herb); p(eff)	occupancy	60.25	0.00	0.32		-34.59		-89.59		17.40
	Ψ(prox+herb); p(.)		60.61	0.36	0.27		-12.52		-18.10		
	Ψ(prox+dog+herb); p(.)		61.07	0.82	0.22		-16.02		-34.87	-8.05	
	occ (prox+herb)	naïve occurence	22.82	0.00	0.47		-20.12		-12.80		
	occ (prox+dog+herb)		24.51	1.69	0.20		-20.65		-16.25	-2.31	
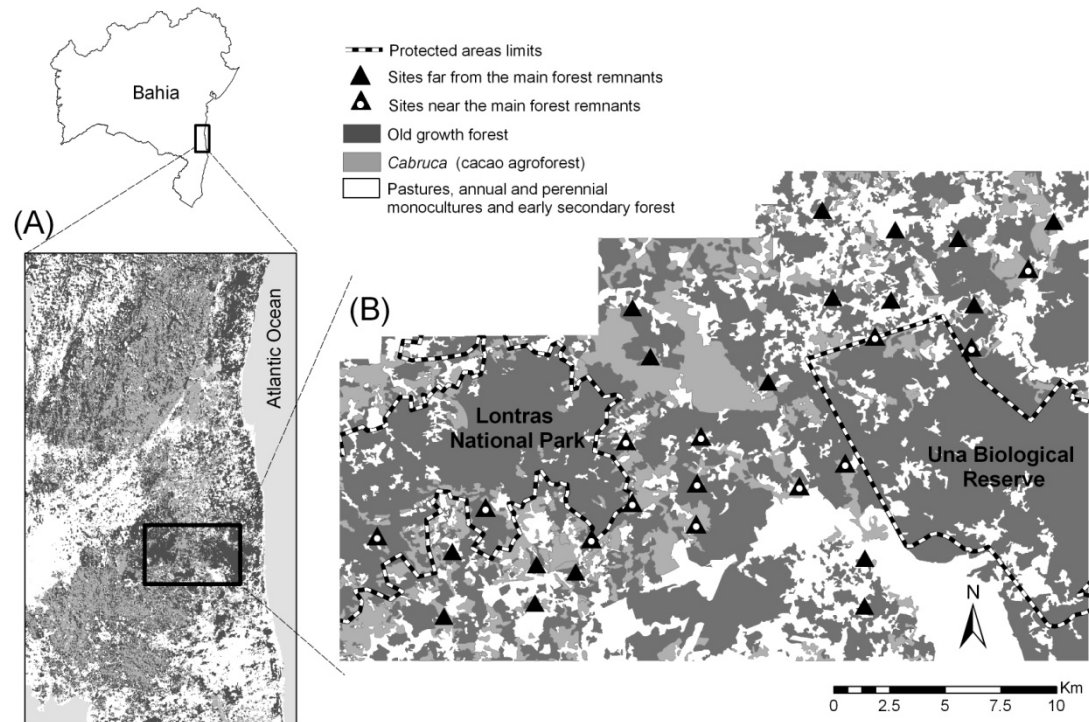
Figure 1: Land-use map of (A) the cacao growing region of southern Bahia and (B) the study region with the location of the study sites near and far from the main forest remnants.

3.6. Supporting Information

Calculation of forest cover variables

Forest cover index: To calculate the forest cover index we used the location of each sampling site as central reference to define circumferences of 400 m radius. We clipped the circumferences from the digital land-use map, divided them into 10 m rings and calculated the amount of forest cover in each ring. Then, we divided the total forest cover within each ring by the average ring distance to the central site and added the 40 values. This procedure gives higher weight to forests near the study site, which are expected to be more important to define the use of the *cabruca* sites by mammals. We chose buffers of 400 m radius over buffers of 800 and 1200 m radius because variability in the amount of forest cover was greater at this scale.

Proximity to large forest remnants: We classified sites as near and far from the main forest remnants if the interior of these remnants were, respectively, included or not in a buffer of 800 m radius around the study site. We defined the main forest remnants as those with forest interior (i.e. area at least 100 m from the edge) larger than 500 ha. This definition of the main forest remnants included those forest patches that are expected to be most important for the populations of most recorded mammals at the study landscape. Remnants of 700 ha were estimated as the minimal patch size to support viable populations of *Leontopithecus chrysomelas* (Zeigler *et al.* 2010), which is the best studied mammal species from our landscape.

Naïve occurrence and count models

We used generalized linear models (GLMs) and mixed models (GLMMs) to model the naïve occurrence (presence/absence) and number of records (number of trap-days the species was recorded) of native mammals in *cabrucas*. We modeled species naïve occurrence using binomial (logit link) GLMs. We controlled for uneven sampling effort on ground and understory stations by excluding extra days from the station with the larger sampling effort before accounting for presence or absence. To control for uneven sampling effort between sites we used it as an explanatory variable and constructed models with and without this variable. We modeled number of records of terrestrial species using negative binomial (log link) GLMs and controlled for uneven sampling effort between sites using its logarithm as an offset argument in the models. For scansorial and arboreal species we

modeled the number of successful recordings (number of records) and failed recordings (number of days the species was not recorded) by trap station strata using quasi-binomial (logit link) GLMMs. In these last models, we informed the study site and trap station strata (ground and understory) as random effect variables. Because the sampling effort corresponds to the sum of successful and failed recordings, we did not include it as an explanatory variable nor as an off-set argument. Negative binomial and quasi-binomial distributions were used to account for overdispersion in our count data.

Three-step procedure used to select occupancy models

We used a three-step procedure to select occupancy models with the variables that best explain the parameters occupancy (Ψ) and detection probability (p). First, we held the proportion of occupied sites constant, $\Psi(.)$, and allowed detection probability to vary, p(var), according to single and multi-variable models. Secondly, we held species detection probability constant, p(.), and modeled occupancy as function of the variables, $\Psi(var)$. For both steps we used a constant model, $\Psi(.)$ p(.), as reference. At each step models were ranked according to Akaike's Information Criterion modified for small samples (AICc; Burnham & Anderson 2002) and values of ΔAICc ≤ 2 indicated equally plausible models. In a third step, the plausible $\Psi(.)$ p(var) and $\Psi(var)$ p(.) models for each species were combined in $\Psi(var)$ p(var) models to explore if including covariates in both parameters improved model performance.

Table S1: Number of candidate occupancy, abundance, naïve occurrence and count models, describing the variables included depending on species locomotor habit (AR = arboreal; SC = scansorial and TE = terrestrial). At the bottom, the number of candidate models ran for each species is shown. In parentheses, the number of occupancy models included in the third step of occupancy model selection (see "Three-step procedure used to select occupancy models" in SI for details).

N° variables	Variables included	Occupancy			Abundance	Naïve occurrence			Count		
		AR	SC	TE	AR	AR	SC	TE	AR	SC	TE
0		1 [a,b]	1 [a,b,c]	1 [a,b,c]	1	1	1	1	1	1	1
1	One forest cover or one management variable	4 [a,b]	5 [a,b]	4 [a,b]	4	4	5	4	4	5	4
2	Combination of: • one forest cover and one management • two management variables	5 [a,b]	9 [a,b]	5 [a,b]	5	5	9	5	5	9	5
3	Combination of: • one forest cover and two management • tree management variables	2 [a,b]	7 [a,b]	2 [a,b]	2	2	7	2	2	7	2
1	Sampling effort	1 [b]	1 [b,c]	1 [b,c]	1	1	1	1			
2	Sampling effort plus: • one forest cover variable • one management variable	4 [b]	5 [b]	4 [b]	4	4	5	4			
3	Sampling effort plus: • one forest cover and one management • two management variable	5 [b]	9 [b]	5 [b]	5	5	9	5			

Table S1 (cont.)

Species	Occupancy			Abundance	Naïve occurrence			Count		
	AR	SC	TE	AR	AR	SC	TE	AR	SC	TE
Callithrix kuhlii				22				12		
Leontopithecus chrysomelas	33(3)			22	22			12		
Didelphis aurita		58(0)				37			22	
Eira barbara		58(16)				37			22	
Leopardus sp.		23(6)							22	
Procyon cancrivorus		58(na)				37			22	
Cabassous sp.			13(2)							12
Cuniculus paca			13(3)							12
Cerdocyon thous			33(2)				22			12

[a] Models run in the first step of occupancy model selection; [b] models run in the second step of occupancy model selection for species recorded in 10 sites or more; [c] models run in the second step of occupancy model selection for species recorded in less than10 sites (see "Three-step procedure used to select occupancy models" in SI for details).

Table S2: Selected models (ΔAICc ≤ 2) describing the variation in the use of 30 *cabruca* sites by large mammals as a function of forest cover, management intensification, and sampling effort. Only shown results of species if the reference model was included among the selected models. For the remaining species see Table 3. Dependant variables: λ = abundance, Ψ = occupancy, p = detection probability, occ = naïve occurrence and rec = number of records. Explanatory variables: fo = forest cover index; prox = proximity to large forest remnants; can = canopy connectivity index; herb = herbaceous vegetation height; dog = domestic dog capture rate; eff = sampling effort. The notation (.) stands for constant.

Species	Model description	Model class	AICc	ΔAICc	w	Forest cover		Management			eff
						fo	prox	can	herb	dog	
Didelphis aurita	Ψ(.); p(.)	occupancy	152.27	0.00	0.13						
	Ψ(.); p(fo)		153.95	1.68	0.06	0.87					
	Ψ(.); p(eff)		154.05	1.78	0.05						1.11
	occ (.)	naïve occurence	38.79	0.00	0.19						
	occ (fo)		40.67	1.87	0.08	1.15					
	occ (eff)		40.72	1.93	0.07						-0.02
	rec (.)	count	145.17	0.00	0.30						
	rec (fo)		147.03	1.86	0.12	1.26					
Eira Barbara	Ψ(prox+can); p(.)	occupancy	96.01	0.00	0.10		-4.36	18.73			
	Ψ(prox+can+herb); p(.)		96.91	0.90	0.06		-35.02	31.29	42.14		
	Ψ (.); p(can)		97.67	1.66	0.04			2.15			
	Ψ (.); p(dog)		97.72	1.71	0.04					-2.19	
	Ψ(.); p(herb)		97.83	1.81	0.04				2.18		
	Ψ(.); p(.)		97.85	1.83	0.04						
	Ψ(dog); p(.)		97.89	1.88	0.04					-2.60	

Table S2 (cont.)

Species	Model description	Model class	AICc	ΔAICc	w	Forest cover		Management			eff
						fo	prox	can	herb	dog	
Eira Barbara	occ (dog)	naïve occurence	41.43	0.00	0.10					-2.13	
	occ (.)		41.57	0.14	0.09						
	occ (prox+can)		42.29	0.86	0.07		-1.45	3.53			
	occ (can)		42.36	0.93	0.06			2.32			
	occ (prox+dog)		42.44	1.02	0.06		-1.01			-2.33	
	occ (prox)		42.67	1.24	0.05		-0.88				
	occ (can+dog)		43.17	1.75	0.04			1.71		-1.83	
	occ(prox+dog+can)		43.39	1.96	0.04		-1.45	2.89		-1.88	
	rec (.)	count	100.15	0.00	0.19						
	rec (dog)		100.31	0.16	0.18					-2.84	
	rec (can)		101.47	1.32	0.10			2.77			
Procyon cancrivorus		occupancy							*no singular result		
	occ (dog)	naïve occurence	42.27	0.00	0.09					-1.97	
	occ (can)		42.46	0.19	0.08			3.52			
	occ (.)		42.52	0.25	0.08						
	occ (fo)		42.94	0.67	0.07	-2.23					
	occ (fo+dog)		43.15	0.88	0.06	-2.23				-1.93	
	occ (can+dog)		43.29	1.02	0.05			2.96		-1.64	
	occ (fo+can)		43.63	1.36	0.05	-1.93		3.06			
	rec (dog)	count	95.70	0.00	0.24					-1.94	
	rec (.)		96.96	1.26	0.13						
	rec (dog+herb)		97.45	1.75	0.10				-1.50	-2.12	
	rec (prox+dog)		97.66	1.96	0.09		-0.60			-1.94	

Table S2 (cont.)

| Species	Model description	Model class	AICc	ΔAICc	w	Forest cover		Management			eff
						fo	prox	can	herb	dog	
Cuniculus paca	Ψ(.); p(.)	occupancy	63.40	0.00	0.16						
	Ψ(.); p(eff)		63.79	0.39	0.13						18.42
	Ψ(fo); p(.)		64.11	0.71	0.11	2.70					
	Ψ(herb); p(.)		64.66	1.27	0.09				2.83		
	Ψ(fo); p(eff)		64.72	1.32	0.08	2.74					18.93
	Ψ(herb); p(eff)		64.97	1.57	0.07				3.34		19.97
	Ψ(fo+herb); p(.)		65.26	1.86	0.06	3.68			3.97		
	occ (.)	naïve occurence	32.17	0.00	0.16						
	occ (fo)		32.51	0.35	0.14	2.63					
	occ (herb)		33.35	1.18	0.09				2.13		
	occ (fo+herb)		33.86	1.70	0.07	2.53			2.33		
	occ (eff)		33.94	1.78	0.07						0.03
	occ (dog)		34.11	1.94	0.06					0.86	
	occ (prox)		34.16	1.99	0.06		0.51				
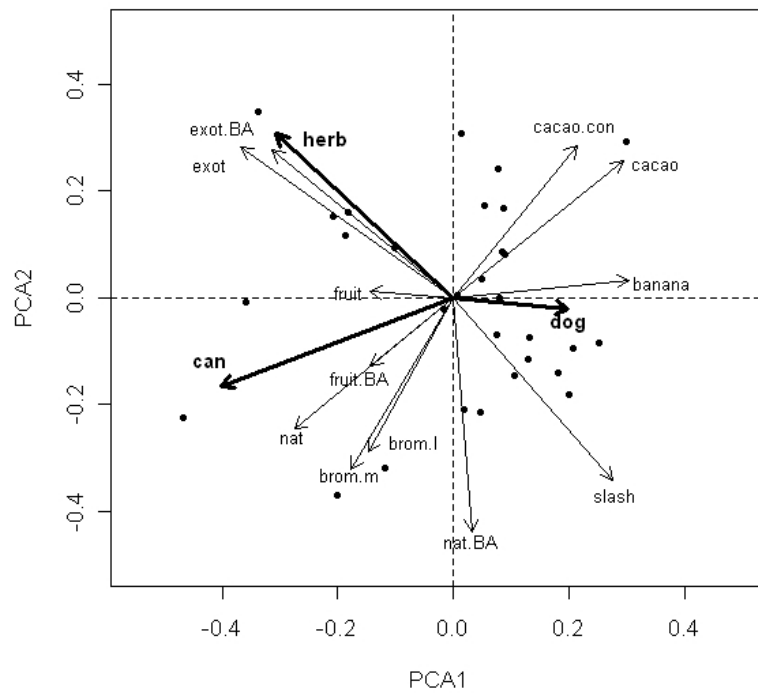
Figure S1: Ordination of sampling sites (dots) in a PCA bi-plot according to vegetation measurements and domestic dog capture rate. Indication of the variables included in model selection analysis (darker arrows): can = canopy connectivity index, herb = average herbaceous vegetation height, and dog = domestic dog capture rate. Other vegetation measurements are: nat, fruit and exot= number of native, exotic fleshy fruit and exotic non-fleshy fruit trees; nat.AB, fruit.AB and exot.AB = sum of basal area from native, exotic fleshy fruit and exotic non-fleshy fruit trees; brom.m and brom.l = number of medium and large bromeliads on tree crows; cacao = number of cacao shrubs; cacao.con = cacao connectivity index; banana = number of banana stems; slash = frequency of herbaceous vegetation clearance.

DISCUSSÃO GERAL

A contribuição de agroflorestas com estrato arbóreo diversificado para a mitigação dos efeitos da perda e fragmentação florestal tem sido apontada em estudos científicos há mais de duas décadas, com avanços recentes no que diz respeito às possibilidades de síntese das similaridades e diferenças nas respostas entre grupos biológicos e à contextualização da paisagem. Perfecto *et al.* (1996), Moguel & Toledo (1999), Rice & Greenberg (2000) discutem a importância das agroflorestas de cacau e café para a conservação de biodiversidade a partir do resultado de estudos, publicados até o final da década de 90, que contrastam as agroflorestas com remanescentes de vegetação nativa e agroecossistemas de manejo mais intensivo. Estas publicações, que sintetizam grande parte do conhecimento sobre a biodiversidade em agroflorestas existente na época, demonstram que poucos estudos haviam abordado a importância da quantidade ou configuração espacial dos ambientes, com grande parte das informações concentradas em poucos grupos biológicos: artrópodes, um grupo de interesse econômico com muitas espécies que atuam como pragas e polinizadores ou são predadores desses, e aves, sobretudo espécies migratórias. As pesquisas realizadas no sul da Bahia e revisadas no capítulo 1 foram, com poucas exceções, publicadas num período mais recente. Além de termos encontrado estudos sobre diversos táxons e sobre espécies ameaçadas de extinção, esta revisão se beneficiou de um conjunto de estudos com delineamento amostral padronizado entre grupos biológicos e da comparação da diversidade de espécies entre duas paisagens com diferentes proporções de cobertura florestal remanescente e agroflorestas. Isto nos possibilitou identificar fatores importantes para manter biodiversidade e recomendar ações de manejo em pelo menos duas escalas distintas: da plantação e da paisagem.

Os capítulos 2 e 3 ampliam o conhecimento sobre a distribuição de mamíferos de maior porte em uma paisagem da região cacaueira do sul da Bahia e para paisagens agroflorestais de maneira geral. As informações sobre mamíferos de maior porte nas *cabrucas* do sul da Bahia disponíveis até 2008 e revisadas no primeiro capítulo são provenientes principalmente de estudos realizados com espécies individuais, e muitos estudos não publicados ou publicados em eventos científicos nacionais, portanto de divulgação limitada. Nesses estudos, as inferências a respeito das respostas dos mamíferos ao tipo de ambiente, intensificação do manejo e redução da cobertura florestal, embora possíveis, são baseadas em evidências fracas devido a conjuntos de dados limitados ou delineamentos

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inadequados para responder tais questões. Para outras regiões Neotropicais, já existiam na época estudos publicados avaliando assembléias de mamíferos de maior porte nas agroflorestas (Estrada *et al.* 1994, Harvey *et al.* 2006), nos quais a inferência sobre a influência do manejo advinha da comparação categórica entre agroflorestas e plantações perenes de manejo intensivo e inferências sobre a importância da cobertura florestal advinham da comparação entre agroflorestas e remanescentes de vegetação nativa. Um único estudo comparando um gradiente de intensificação de plantações de café (Gallina *et al.* 1996) se baseou no contraste de apenas quatro sítios amostrais. A comparação entre *cabrucas* e florestas, realizada no capítulo 2, teve como diferencial em relação a esses trabalhos anteriores um delineamento pareado com o qual minimizamos a influência do contexto da paisagem no entorno dos sítios sobre os padrões observados. No capítulo 3, além de considerarmos o manejo das agroflorestas ao longo de um gradiente com 30 unidades amostrais, comparamos a importância relativa da intensificação do manejo e da cobertura florestal no entorno dos sítios.

Assim como avanços importantes aconteceram nos estudos realizados no sul da Bahia, o conhecimento sobre biodiversidade em sistemas agroflorestais tem sido ampliado em outras regiões onde esses sistemas ocupam grandes extensões da matriz, com comparações multi-taxa cada vez mais freqüentes (ex.: Perfecto *et al.* 2003, Steffan-Dewenter *et al.* 2007, Clough *et al.* 2011, Waltert *et al.* 2011) e estudos contrastando a importância do manejo versus a distribuição de cobertura vegetal remanescente (ex: Bali *et al.* 2007, Clough *et al.* 2009, Clough *et al.* 2010). Além disso, estudos avaliando grupos com diferentes requerimento de habitat têm apontado as limitações das agroflorestas em manter espécies estritamente florestais, tanto na Bahia (Pardini *et al.* 2009) quanto fora dessa região (Waltert *et al.* 2011), e a avaliação de grupos funcionais tem permitido inferências sobre a importância da biodiversidade das agroflorestas na manutenção de serviços ecossistêmicos (Klein *et al.* 2003, Clough *et al.* 2009).

Principais resultados e implicações para a conservação de biodiversidade em paisagens agroflorestais

De maneira similar às revisões de trabalhos publicados até o final da década de 90, os estudos compilados no capítulo 1 demonstram que as agroflorestas com sombreamento diversificado são utilizadas por uma parcela substancial da fauna e flora nativas, mas que

isso depende de características da plantação que são, por sua vez, afetadas pelo manejo. Ao rever elementos chave que influenciam a diversidade das comunidades e o uso das *cabrucas* por espécies ameaçadas de extinção, tecemos recomendações para o manejo das plantações que vão desde a indicação de espécies arbóreas para serem mantidas através do planejamento de remoção, poda e regeneração, até a indicação de estruturas simples que aumentam a complexidade estrutural (ex: troncos mortos, cupinzeiros e epífitas) e a coibição da caça.

No que se refere ao manejo na escala da paisagem, sugerimos, em paisagens muito desmatadas, o acréscimo de cobertura vegetal atendendo à legislação ambiental vigente e um manejo das agroflorestas "amigável" à conservação. Essas ações devem garantir grande quantidade de habitat para as espécies menos vulneráveis e a manutenção de metapopulações viáveis de espécies com distribuição limitada nas *cabrucas*. Já em paisagens com porcentagem de cobertura florestal elevada (tomando uma paisagem com 50% de cobertura florestal como referência), os estudos revistos indicam que as agroflorestas ampliam a conectividade e garantem uma distribuição bastante homogênea das espécies. Nessas paisagens, remanescentes florestais pequenos mantêm assembléias de espécies similares aos trechos de vegetação contínua e esta parece ser uma situação favorável para a manutenção de populações da maior parte das espécies nativas, com exceção daquelas diretamente exploradas pelo homem como mamíferos (capítulos 2 e 3) e árvores (Pardini *et al.* 2009). Partindo das semelhanças de composição e estrutura registradas entre as *cabrucas* e os remanescentes florestais, sugerimos ainda o uso das agroflorestas como buffers no entorno de remanescentes florestais para reduzir efeitos de borda.

Nos capítulos 2 e 3 mostramos que, embora o número de espécies registradas nas *cabrucas* e remanescentes florestais de um mosaico agroflorestal com grande porcentagem de vegetação nativa seja elevado, a representatividade de espécies caçadas ou mortas por retaliação é baixa quando comparada a estudos que utilizaram o mesmo método de amostragem em florestas contínuas (ex: Silveira *et al.* 2003, Srbek-Araujo & Chiarello 2005, Tobler *et al.* 2008). Também mostramos que a maior incidência de cães domésticos, considerada uma consequência indireta da intensificação do manejo, reduz o uso das *cabrucas* por mamíferos e que tanto a conversão de florestas em *cabrucas* quanto a intensificação do manejo das agrofloretas afetam negativamente espécies arborícolas e positivamente espécies generalistas. Ao comparar as causas proximais (intensificação do

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manejo versus cobertura florestal) responsáveis pela redução do uso das *cabrucas* por mamíferos de maior porte, encontramos que as variáveis relacionadas ao manejo têm efeitos negativos mais importantes sobre este grupo. Argumentamos, no entanto, que este resultado deve ser entendido dentro do contexto da área de estudo - uma paisagem bastante florestada e com uma assembléia de mamíferos empobrecida em conseqüência da sobrecaça - e que, a exemplo de outros estudos (ex: Tscharntke *et al.* 2005, Pardini *et al.* 2010, Batary *et al.* 2011), respostas mais pronunciadas à intensificação do manejo e/ou à redução da cobertura florestal numa escala pequena devem ocorrer em paisagens mais desmatadas, alterando a importância relativa entre esses fatores.

Diante desses resultados, inferimos que o controle de populações de cães domésticos e de atividades de caça são as ações de manejo que devem produzir grande benefício para a manutenção de mamíferos de maior porte em paisagens agroflorestais e podem ser implementadas sem reduzir a produtividade local das plantações. A manutenção de sombreamento deve ser adotada com o objetivo de aumentar a disponibilidade de habitat e conectividade da paisagem para espécies arborícolas, mas deve ser avaliada também em função da produtividade. Na paisagem de estudo, a elevada porcentagem de cobertura florestal é importante e deve ser mantida, porém a distribuição espacial dos remanescentes exerce pouca influência nas populações de mamíferos de maior porte.

Desafios futuros para aprimorar o uso das agroflorestas como ferramenta para conservação

Após reunir informações sobre a biodiversidade nas *cabrucas* do sul da Bahia no capítulo 1, identificamos questões que, ao serem investigadas e esclarecidas por estudos futuros, poderão auxiliar no refinamento das estratégias para conciliar a conservação e a produção agrícola nas paisagens agroflorestais. Entre os assuntos identificados como prioritários e carentes de conhecimento estão: (1) densidades de sombreamento "ótimas" que maximizem ganhos ambientais e econômicos, (2) limiares de cobertura florestal e fragmentação relacionados à perda acentuada de espécies em paisagens agroflorestais e (3) identificação de relações entre recursos chave (ex. disponibilidade de alimento e abrigo, estrutura da vegetação, condições microclimáticas) e a diversidade e riqueza de espécies nas *cabrucas*, principalmente espécies dependentes de floresta.

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Relações entre produtividade e diversidade de espécies têm sido avaliadas em plantações de cacau da Indonésia. Clough *et al.* (2011) encontraram uma riqueza constante de pequenos vertebrados, invertebrados, fungos e árvores ao longo de um gradiente de produtividade entre plantações, mesmo considerando apenas espécies florestais no caso das aves. Para esses autores, níveis moderados de sombreamento (30-40%) e tratos manuais da lavoura podem garantir produtividade sem reduzir o valor de conservação das agroflorestas, ao menos em regiões tropicais ocupadas por pequenas propriedades. A partir disso, sugerem que estratégias *wildlife-friendly* podem resultar nos maiores ganhos ambientais e econômicos nessas paisagens. Em um trabalho anterior, Steffan-Dewenter *et al.* (2007) também sugerem um máximo benefício ambiental e econômico em níveis intermediários de sombreamento (35-50%) ao contrastar o manejo local com a riqueza de espécies (invertebrados e plantas) e estimativas de serviços ecossistêmicos. Porém, para esses autores esta estratégia é uma opção intermediária entre a produção extensiva (*wildlife-friendly*) em larga escala e a intensificação das plantações aliada à proteção integral de áreas maiores de floresta. Os autores pontuam ainda alguns fatores importantes no que se refere ao balanço entre intensificação do manejo e manutenção de cobertura florestal nas estratégias para conciliar produtividade e conservação, com os quais chamam atenção para questões mais abrangentes que não podem ser respondidas em seu estudo. Primeiro, apontam a ausência de informação sobre efeitos adicionais aos do manejo que podem decorrer do isolamento das plantações em relação a áreas fonte, ou seja, que o nível de sombreamento "ótimo" encontrado por eles pode variar entre paisagens com diferentes porcentagens de cobertura florestal e em função da distribuição espacial dos remanescentes de vegetação. Em segundo lugar, indicam que poucos pequenos produtores rurais possuem capacidade de sustentar altos níveis de produtividade nas lavouras intensificadas devido ao alto custo de produção, portanto, que o componente sócio-econômico impõe limitações à intensificação. Finalmente, destacam a importância de se definir metas de conservação: serviços ecossistêmicos associados à biodiversidade ou táxons ameaçados altamente dependentes dos habitats naturais?

De fato, grande parte das publicações a respeito de estratégias para conciliar produção agrícola e conservação analisa apenas parte das questões envolvidas, e isso parece limitar as conclusões a respeito da contribuição da intensificação do uso da terra e manutenção de habitat nativo. Entre os estudos que apontam a intensificação agrícola associada a mais áreas com vegetação intacta como melhor alternativa, uma premissa comum é de que

existe uma relação direta entre a extensão de cobertura de vegetação remanescente e conservação de biodiversidade (Balmford *et al.* 2005, Ewers *et al.* 2009, Phalan *et al.* 2011), ignorando a importância de processos ecológicos que dependem da matriz agrícola e de distúrbios que se propagam desta para os remanescentes de vegetação. Já os estudos que apontam os sistemas extensivos como opção mais favorável raramente consideram uma possível expansão das plantações, decorrente do aumento da demanda, e seu avanço sobre áreas de vegetação nativa (Harvey *et al.* 2008, Perfecto & Vandermeer 2008).

No que se refere ao conhecimento biológico, esta discussão é limitada em parte porque os estudos nos quais se baseiam avaliaram a distribuição das espécies ou assembléias tendo sítios ou manchas de ambientes (ex. fragmentos, plantações) como unidades amostrais. Segundo Bennett *et al.* (2006), a compreensão das propriedades de mosaicos de ambientes devem ter como unidades de amostragem mosaicos inteiros, com seus diversos usos da terra representados. Entretanto, vale ressaltar que a escala de resposta das espécies varia. Enquanto alguns quilômetros quadrados podem ser considerados unidades amostrais válidas para representar mosaicos de uso da terra que influenciam na dinâmica de populações de algumas espécies (Bennett *et al.* 2006), nosso estudo indica que mamíferos de maior porte devem responder a características da paisagem em escalas muito mais amplas. O papel das agroflorestas para conciliar conservação e produção agrícola será aprimorado também com o desenvolvimento de mais estudos que quantifiquem, além da diversidade de espécies, os processos ecológicos mantidos nas plantações, bem como a influência desses sistemas na atenuação de efeitos de borda, que constitui um fator importante para a deterioração do habitat florestal em paisagens fragmentadas (Oliveira *et al.* 2008, Faria *et al.* 2009).

Como mencionado nas discussões dos capítulos, o conhecimento atual permite a inferência de medidas para garantir e/ou aumentar a diversidade biológica em paisagens modificadas pelo homem e pode ser traduzido em ações de manejo. No que diz respeito ao embasamento biológico, a avaliação e adaptação das práticas sugeridas com o avanço do conhecimento são pontos importantes para garantir a sustentabilidade dessas paisagens. Outros desafios tão ou mais importantes para viabilizar e manter no longo prazo a biodiversidade e serviços ambientais associados e a produção agrícola são a incorporação de fatores sócio-econômicos e o suporte de políticas públicas que, por sua vez, dependem do envolvimento de profissionais de áreas distintas e da conciliação de interesses de diferentes segmentos da sociedade.

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