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CACHORROS VS MESOCARNÍVOROS NATIVOS: INTERAÇÕES ESPAÇO-TEMPORAIS EM PAISAGENS AGROFLORESTAIS

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Cachorros vs mesocarnívoros nativos: interações espaço-temporais em paisagens agroflorestais

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I. INTRODUÇÃO GERAL

Ações humanas sobre os organismos e seus habitats vem alterando a fauna silvestre há centenas de anos, com efeitos mais pronunciados nas últimas décadas (Dirzo et al. 2014). As ameaças decorrentes das ações antrópicas, como a perda e fragmentação do habitat, mudanças no tipo de uso de solo, presença de espécies invasoras e a caça, estão inter-relacionadas (Pardini et al. 2017). Por exemplo, mudanças do habitat pode aumentar a abundância de espécies invasoras, possivelmente levando a um maior impacto nas espécies nativas em áreas perturbadas (Didham et al. 2007). Pequenos fragmentos florestais em paisagens antropizadas, são exemplos de áreas perturbadas cujo valor de conservação depende de características da paisagem nas quais estão inseridos (Pardini et al. 2017). Esses fragmentos florestais geralmente estão inseridos em uma matriz predominantemente destinada à agricultura e/ou pecuária (Ribeiro et al. 2009). As matrizes podem ser permeáveis ou impermeáveis para diferentes espécies, determinando a conectividade entre fragmentos de habitat, ou podem funcionar como habitats secundários, dependendo das características ecológicas próprias das espécies (Boscolo et al. 2016). Por se tratar de ambientes, condições e recursos diferentes do habitat original, as matrizes tendem a favorecer espécies invasoras (Kupfer et al. 2006).

Segundo Simberloff and Rejmánek (2011), as espécies invasoras podem ser definidas por dois conceitos complementares. O primeiro, e mais aceito na ecologia, é: “uma espécie naturalizada que produz descendentes reprodutivos, frequentemente em números muito grandes, e que se espalha em grandes áreas”, enquanto para os conservacionistas são “espécies não nativas que se propagam rapidamente, causando danos ambientais ou econômicos”. Esta segunda definição destaca que as espécies invasoras tem um impacto negativo nas comunidades de espécies nativas, seja por competição direta, indireta ou aparente, por predação, transmissão de patógenos, modificação de habitat ou alteração dos ciclos de nutrientes e água (da Rosa et al. 2017).

Uma das espécies invasoras mais comuns no mundo é o cachorro doméstico (*Canis familiaris*). Dada sua relação com as pessoas, estes animais estão presentes em praticamente todas as regiões do globo (Daniels and Bekoff 1989). De acordo com a relação que têm com o ser humano e com o tipo e forma de circulação no ambiente utilizado, os cachorros podem ser categorizados quatro classes principais: a) cães domiciliados, aqueles que possuem estreito vínculo social com o ser humano e com locomoção restrita aos limites ou intermediações da propriedade de seu dono; b) cães urbanos de vida livre, aqueles cães de rua ou errantes que não são contidos por seres humanos e vagueiam livremente pelas ruas, porem precisam dos restos alimentares de humanos para sobreviver; c) cães rurais de vida livre, aqueles cães rurais semi-errantes ou semi-domiciliados que são subsidiados por dono, mas não possuem barreiras físicas que restringem sua área de vida e podem circular livremente por ambientes naturais; e d) cães ferais, os quais são asselvajados, sem qualquer tipo de vínculo social com o ser humano, vivendo agrupados em matilhas e não dependem de subsídio alimentar dos humanos (Vanak and Gompper 2009).

Os cachorros de vida livre e os ferais que compartilham ambientes naturais ou semi-naturais com a fauna nativa podem atuar como predadores, reservatório de doenças e competidores por recursos (Vanak and Gompper 2009). Além disso, os cães rurais de vida livre são frequentemente usados pelas pessoas para caçar, e seu comportamento e impacto na fauna estão influenciados pelos humanos (Castilho et al. 2017; dos Santos et al. 2018). Portanto, os cachorros podem afetar negativamente as populações de mamíferos nativos (Doherty et al. 2017), alterando as interações interespecíficas entre elas e influenciando na estrutura das comunidades (Farris et al. 2015; Zapata-Ríos and Branch 2018).

As interações interespecíficas são essenciais nas dinâmicas das comunidades dos ecossistemas e podem atuar como filtros ecológicos para determinar quais espécies podem coexistir (Schoener 1974; Linnell and Strand 2000). Para que espécies simpátricas com nicho ecológico semelhantes coexistam, a partição de recursos é um mecanismo fundamental de coexistência. Nesta partição de recursos existem três mecanismos principais nos quais as espécies podem se diferenciar: pela dieta (alimentando-se de diferentes itens), no espaço (priorizando o uso de diferentes tipos de ambientes ou território) e no tempo (estando ativos em diferentes horas no ciclo diário) (Schoener 1974). Os mamíferos carnívoros (ordem Carnívora, Mammalia) são um exemplo de espécies simpátricas com sobreposição de nicho ecológico. Dos três mecanismos acima mencionados, a dieta é a mais estudada na co-ocorrência dos carnívoros, seguida pela segregação espacial e temporal (Davis et al. 2018).

Esta partição dos recursos entre carnívoros, pode ser afetada negativamente pela introdução de outros carnívoros invasores, como os cachorros (Farris et al. 2015; Zapata-Ríos and Branch 2016). Em várias regiões do mundo, os cachorros são considerados como a espécie dominante na assembleia de mesocarnívoros (espécies da ordem Carnivora de nível trófico médio que pesam até 15 kg; Prugh et al., 2009), devido a sua alta abundância, ampla distribuição espacial, maior tamanho e a sua relação com o humano (Vanak and Gompper 2010; Vanak et al. 2016).

A coexistência entre cachorros e mesocarnívoros nativos tendem a ser intensas em ambientes semi-naturais, tais como bordas de florestas (Paschoal et al. 2012, 2016) e sistemas agroflorestais (Cassano et al. 2012, 2014). Isso acontece pelo fato desses ambientes semi-naturais ainda manterem uma considerável diversidade de espécies nativas quando comparado com outros ambientes mais perturbados (Bali et al. 2007; Caudill et al. 2014a), porém são mais sujeitos a invasão pelos cachorros, que acessam as florestas a partir das bordas e entram nos sistemas agroflorestais junto com os trabalhadores (Frigeri et al. 2014; dos Santos et al. 2018).

Uma região ideal para estudar o impacto dos cachorros nos mamíferos nativos é a Mata Atlântica do sul da Bahia. Esta região é atualmente ocupada por um mosaico agroflorestal constituído principalmente por agroflorestas de cacau (*Theobroma cacao*), fragmentos florestais e áreas abertas (Faria et al. 2007; Pardini et al. 2009). Apesar das perturbações humanas que sofre e a alta presença de cachorros, estas paisagens agroflorestais ainda abrigam várias espécies de mamíferos de médio e grande porte (Cassano et al. 2012). No entanto, é importante ressaltar a ausência ou escassos registros de predadores de topo, como a onça pintada *Panthera onca* e a onça parda *Puma concolor* (Moura 2003; Cassano et al. 2012). Neste cenário, os mesocarnívoros podem atuar como um substituto de predadores de topo e desempenhar importantes funções ecológicas no ecossistema (Prugh et al. 2009).

Diferentes estudos avaliam o efeito das espécies exóticas na ocupação, probabilidade de detecção ou nos padrões de atividade dos mamíferos nativos (e.g. Gerber et al. 2012; Farris et al. 2014, 2015; Wang et al. 2015; Bogdan et al. 2016; Zapata-Ríos and Branch 2016, 2018). Para isso, utilizam armadilhas fotográficas, modelos de ocupação (Mackenzie et al. 2002) e o coeficiente de sobreposição temporal de Ridout e Linkie

(2009). Está comprovado que as armadilhas fotográficas são uma ótima ferramenta para estudar espécies elusivas e crípticas, a exemplo dos carnívoros. Esta é uma técnica não invasiva, com capacidade para registro dos organismos durante 24 horas por dia, permitindo a identificação da espécie, hora e data do registro (Rovero and Zimmermann 2016). Tais informações permitem uma variedade de análises, incluindo a construção de modelos de ocupação (Mackenzie et al. 2002) e a descrição de sobreposição temporal dos padrões de atividade (Frey et al. 2017).

Os modelos de ocupação propostos por Makenzie e colaboradores (2002) permitem avaliar os padrões espaciais da ocorrência das espécies, fornecendo estimativas que diminuem o viés da detecção imperfeita (<1). A ocupação é definida como a proporção de área, fragmentos ou locais ocupados por uma espécie (Mackenzie et al. 2002). As bases dos modelos de ocupação são dados de presença (1) e ausência (0) de uma espécie por um determinado período de amostragem, assim como a informação de covariáveis que também possam influenciar na probabilidade de ocupação e detecção. Quando um estudo é desenvolvido em vários anos ou estações (como nesta dissertação), é recomendável utilizar o "multi-season occupancy modeling", porque além da probabilidade de ocupação e detecção, também considera as probabilidades de extinção e colonização locais que podem acontecer ao longo do tempo (MacKenzie et al. 2003).

Os padrões de atividade referem-se ao momento em que uma espécie está ativa para desempenhar suas funções ecológicas em um ciclo de 24 horas (Frey et al. 2017). Atualmente, em pesquisas com armadilhas fotográficas, se utiliza a estimativa de densidade de Kernel, que considera cada registro fotográfico de uma espécie como uma amostra aleatória de uma distribuição contínua, ao invés de agrupar registros em categorias de tempo arbitrariamente predefinidas (Ridout and Linkie 2009). Assim, é possível determinar a atividade temporal das espécies, e estimar o nível de sobreposição temporal de duas espécies ou da mesma espécie em diferentes ambientes (Ridout and Linkie 2009).

O objetivo desta dissertação é estudar a influência de cachoros rurais de vida livre sobre os padrões espaço-temporais de mesocarnívoros nativos, em agroflorestas de cacau de três paisagens com diferentes quantidades de cobertura florestal e de agrofloresta, alem de uma área controle constituída por grandes remanescentes florestais. Especificamente pretendemos a) compreender se a frequencia de registros dos cachorros, tipo de ambiente (agroforesta de cacau e remanescentes florestais) ou os diferentes paisagens influenciam na probabilidade de ocupação e detecção de mesocarnívoros nativoss, e b) investigar os padrões de atividade temporal dos mesocarnívoros nativos nas três paisagens e área controle, relacionando com a frequência de registros e período de atividade dos cachorros.

Esta dissertação é composta por um único capítulo que foi escrito em forma de artigo científico, para ser submetido à revista Journal of Mammalogy.

II. Artigo

Free-ranging dogs vs. native mesocarnivores: spatio-temporal interactions in agroforestry landscapes

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Abstract

Interspecific interactions are essential for the dynamics of ecosystem communities, acting as ecological filters to determine species co-occurrence. However, these interactions could be negatively impacted by land use changes, hunting, and the presence of exotic species. The domestic dog (*Canis familiaris*) is the most abundant exotic mammal in human modified landscapes; and cacao agroforestry systems in the Atlantic Forest are unique scenarios to study its influence on native mammals. Because native mesocarnivores share taxonomic and ecological similarities with dogs (which are also used for hunting), they may be negatively impacted by this invasive species. With camera traps, we surveyed for seven years cacao agroforests (45 sites) from three landscapes with different amounts of forest cover and one control area (15 sites) of forest remnants. Multi-season occupancy modeling and activity patterns analysis were used to evaluate the influence of free-ranging dogs on the spatio-temporal patterns of the native mesocarnivores. We found that dogs negatively influenced the spatial or temporal patterns of most native mesocarnivores. Crab-eating fox (*Cerdocyon thous*) and South American coati (*Nasua nasua*) temporally avoided dogs ($p=0.04$ and $p<0.001$, respectively) that also negatively influenced in the detection probability of *N. nasua* (AICc Weight=0.73, $beta_{dogs}= -0.11$, SE=0.05, $-0.21 \leq IC\ 95\% \leq -0.01$). Dogs may negatively influence the occupancy of tayra (*Eira barbara*, AICc Weight=0.66,), but not its activity patterns. However, this negative effect on tayras occupancy is uncertain ($beta_{dogs}=-0.97$, SE=0.59, $-2.12 \leq IC\ 95\% \leq 0.18$). Finally, we did not find evidence for a dog impact on crab-eating raccoon (*Procyon cancrivorus*). These results confirm the impact of dogs in wildlife, and can be extrapolated to similar agroecosystems, and possibly around protected areas. Thus, actions to control free-ranging dog populations (i.e. sterilization and removal campaigns) should occur in agroforestry landscapes and around native forests, as well education programs focusing on responsible dog ownership.

Key words: Activity patterns, Atlantic Forest, cacao agroforestry, dogs, carnivores, interspecific interactions, exotic species, occupancy model.

Resumo

Interações interespecíficas são essenciais para a dinâmica das comunidades ecossistêmicas, atuando como filtros ecológicos para determinar a co-ocorrência de espécies. No entanto, essas interações podem ser afetadas negativamente pelas mudanças no uso da terra, caça e a presença de espécies exóticas. O cachorro (*Canis familiaris*) é o mamífero exótico mais abundante em paisagens humanas modificadas; e sistemas agroflorestais de cacau na Mata Atlântica são cenários únicos para estudar sua influência em mamíferos nativos. Como os mesocarnívoros nativos compartilham semelhanças taxonômicas e ecológicas com os cachorros (que também são usados para a caça), eles podem ser os mais afetados por sua presença. Pesquisamos agroflorestas de cacau (45 locais) de três paisagens com diferentes quantidades de cobertura florestal e uma área de controle (15 locais) de remanescentes florestais usando armadilhas fotográficas. Modelos de ocupação de multi-temporal e análise de padrões de atividade foram usados para avaliar a influência dos cachorros de vida livre sobre os padrões espaço-temporal de mesocarnívoros nativos. Descobrimos que os cachorros influenciaram negativamente os padrões espaciais ou temporais da maioria dos mesocarnívoros nativos. A raposa (*Cerdocyon thous*) e o quati (*Nasua nasua*) evitaram os cachorros temporalmente, e os cachorros também influenciaram negativamente na probabilidade de detecção de *N. nasua*. Além disso, os cachorros influenciaram negativamente a ocupação da irara (*Eira barbara*), mas não seus padrões de atividade. Finalmente, não encontramos evidências de um impacto dos cachorros sobre o guaxinim (*Procyon cancrivorus*). Estes resultados confirmam o impacto dos cachoros na fauna nativa e podem ser extrapolados para outros tipos similares de agroecossistemas, e possivelmente em torno de unidades de conservação. Assim, ações para controlar populações de cachorro (quer dizer, campanhas de esterilização e remoção) devem acontecer em paisagens agroflorestais e em torno de florestas nativas, assim como também como programas de educação com foco na posse responsável de cachorros.

Palavras-chave: Agrofloresta de cacau, cachorros, interações interespecíficas, Mata Atlântica, mesocarnívoros, modelo de ocupação, padrões de atividade.

Introduction

Interspecific interactions exert great influence on the communities structure, facilitating or limiting the survival, growth and reproductive success of the species, and thus interfering with the abundance and viability of populations (Linnell and Strand 2000). For sympatric species with similar ecological niche, resource partitioning in diet, space and time are essential mechanisms to reduce competitive interactions and allow them to co-occur in a same landscape (Schoener 1974). Carnivorous (Order Carnivora, Mammalia) assemblages are an example of sympatric species with niche overlap. Within this group, subordinate species might evade activity centers or areas of higher population density of a dominant species (Linnell and Strand 2000), while other species may adapt their diel activity patterns to reduce temporal activity overlap with a dominant competitor or predator (Bischof et al. 2014; Zapata-Ríos and Branch 2016).

The resource partitioning among medium and large-sized mammals can be affected by several anthropogenic factors, such as the reduction of forest cover because land use changes, hunting, and the presence of exotic species (Schuette et al. 2013; Wang et al. 2015; Parsons et al. 2016). For instance, different amount of forest cover in agroecosystem landscapes influence the occurrence of mammalian carnivores in the Brazilian Atlantic Forest (Regolin et al. 2017), hunting reduced the diurnal activity patterns and number of records of several mammals in an Ecuadorian rainforest (Blake et al. 2013), and exotic carnivores (*e.g.* domestic dogs) reduce the site use of several native carnivores in Madagascar (Farris et al. 2015).

Dogs (*Canis familiaris*) are among the most widespread exotic species, causing several negative impacts on wildlife (Doherty et al. 2017) and decreasing the fitness of native species, through competition, predation, and diseases transmission (Vanak and Gompper 2009). Moreover, dogs are used by people for hunting wildlife and are considered a severe threat for mammals species (Koster 2009). Several studies concluded that the presence of dogs negatively influence on the distribution, occupancy, and activity patterns of native mammals (Silva-Rodríguez et al. 2010; Gerber et al. 2012; Silva-Rodríguez and Sieving 2012; Farris et al. 2014, 2015; Moreira-Arce et al. 2015; Zapata-Ríos and Branch 2016, 2018) using occupancy models (Mackenzie et al. 2002) and temporal overlap analysis (Ridout and Linkie 2009). In the Neotropics, these influence have been mainly addressed in protected areas (Srbek-Araujo and Chiarello 2008; Lacerda et al. 2009; Massara et al. 2015; Lessa et al. 2016) and in rural landscapes with native forest, exotic forest plantations, monoculture croplands, and pasture (Campos et al. 2007; Silva-Rodríguez et al. 2010; Martinez et al. 2013). However, little attention has received the agroforestry landscapes, that are unique scenarios to study the interaction among exotic and native species, because harbor high wildlife diversity and free-ranging dogs are very common (Cassano et al. 2012, 2014).

In cacao (*Theobroma cacao*) agroforestry landscapes of the Atlantic Forest of southern Bahia State, Brazil, increasing rates of free-ranging dogs records have a stronger negative effect on the occurrence of native mammals than the amount of forest cover or local vegetation structure (Cassano et al. 2014). Moreover, in this region dogs are: a) among the most abundant carnivores (Cassano et al. 2014), b) are more active during daylight and hours when people are working (Frigeri et al. 2014; dos Santos et al. 2017), c) are subsidized by people, whose determine its impacts on wildlife (dos Santos et al. 2018), and d) are used for hunting and persecuting wildlife (Castilho et al. 2017). For all these statements, dogs can be considered a dominant species, capable of changing native mesocarnivores (i.e. carnivore species situated at an intermediary trophic level that could weigh until 15 kg) (Prugh et al. 2009) distribution and likely to decrease diurnal activity of species within this group.

In the Atlantic Forest of southern Bahia State, larger carnivores as jaguars (*Panthera once*) are nearly to be locally extinct and puma (*Puma concolor*) records are rare (Moura 2003; Canale et al. 2012; Cassano et al. 2014). Thus, in the study area, the carnivore assemblage is mainly composed by at least eight native mesocarnivore species: the crab-eating fox (*Cerdocyon thous*), crab-eating raccoon (*Procyon cancrivorus)*, coati (*Nasua nasua)*, tayra (*Eira barbara)*, kinkaju (*Potos flavus*), margay (*Leopardus wiedii*), ocelot (*Leopardus pardalis*), and southern tiger cat (*Leopardus guttulus*) (Cassano et al. 2014; Dechner et al. 2018).

Mesocarnivores response to landscape composition, forest cover, type of environment, and to the presence of dogs will depend on its habitat specialization (i.e. habitat generalist species or forest-dependent species), competition (for space, time, and food), and hunting pressure (Cassano et al. 2014; Lessa et al. 2016; Massara et al. 2016). Generally, *C. thous* and *P. cancrivorus* are more commonly associated to less structured forest (Michalski et al. 2006; Regolin et al. 2017) while the *E. barbara, N. nasua* and *Leopardus spp.* are usually forest-dependent species (Ford and Hoffmann 1988; Gompper and Decker 1998; Oliveira 1998; Regolin et al. 2017). *C. thous, P. cancrivorus* and *Leopardus spp.* are mainly nocturnal species, while the *E. barbara* and *N. nasua* are mainly diurnal (Gonzalez-Maya et al., 2009; Massara et al., 2016). Regarding its food habits, *C. thous, P. cancrivorus, E. barbara* and *N. nasua* are omnivorous, while *Leopardus spp.* feed mainly on small vertebrates (Presley 2000; Rocha-Mendes et al. 2010; Bianchi et al. 2014). Moreover, *N. nasua* and *Leopardus spp.* are hunted by local people using dogs, whose influence its behavior and impacts on wildlife (Castilho et al. 2017; dos Santos et al. 2018).

We aim to study the influence of free-ranging dogs on the spatio-temporal patterns of native mesocarnivores in cacao agroforest from three broad landscapes with different amounts of forest cover and in one control area formed by large forest remnants. Specifically, we aim to a) understand whether the records of dogs, type of environment (cacao agroforest or forest remnants) or landscape influence in the occupancy and detection probabilities of native mesocarnivores; and b) determine temporal activity patterns of native mesocarnivores in cacao agroforestry landscapes and forest remnants, relating to the frequency of records and activity period of dogs. For the first objective, we expect that the species that would be less affected by dogs are those that have a similar ecological niche with dogs, as the habitat generalist *Cerdocyon thous* and *Procyon cancrivorus*; while the more affected would be the forest-dependent species: *Nasua nasua*, *Eira barbara* and *Leopardus spp*. On the other hand, the most affected species could also be those ones that are hunted and persecuted with dogs for consumption or retaliation: *N. nasua* and *Leopardus spp*. For the second objective, we expect that diurnal species (*N. nasua* and *E. barbara*) would have a higher shift on its temporal activity patterns to avoid dog activity (also diurnal) in cacao agroforest, while the nocturnal species (*C. thous, P. cancrivorus,* and *Leopardus spp.)* would have less or not shift on its activity patterns.

Material and Methods

2.1 Study area and sampling design

The study was conducted in southern Bahia state in Brazil, a region within the Atlantic Forest, one of the top biodiversity hotspots in the world, given the high level of habitat loss, fragmentation, and the high species richness (Myers et al. 2000). The average annual temperature is 24°C and the mean annual rainfall is 1500 mm, without a distinct dry season, although a warmer and rainless period may occur between December and March (Mori et al. 1983; Thomas et al. 1998).

We established 45 sampling sites in cacao agroforest from three agroforestry landscapes with different amount of cacao agroforest and forest cover, as well in 15 sampling sites in a control area (large forest remnants). All sampling sites were placed at least 1.5 km apart from each other, as several camera trap studies for medium-sized mammals (Burton et al. 2015). The border of each agroforestry landscape and control area was calculated by creating a 1.5 km buffer around the polygon minimum convex that include all the camera traps (Figure 1). We chose this measure to be the same distance as the minimum distance among the cameras. Within these areas, we calculate using ArcGis 10.4 (ESRI, 2016) the percentage of cacao agroforest, forest, open areas, and *Ecualyptus* plantations in each landscape, by using the land use classification from the project REDE SISBIOTA (Morante-Filho et al. 2015).

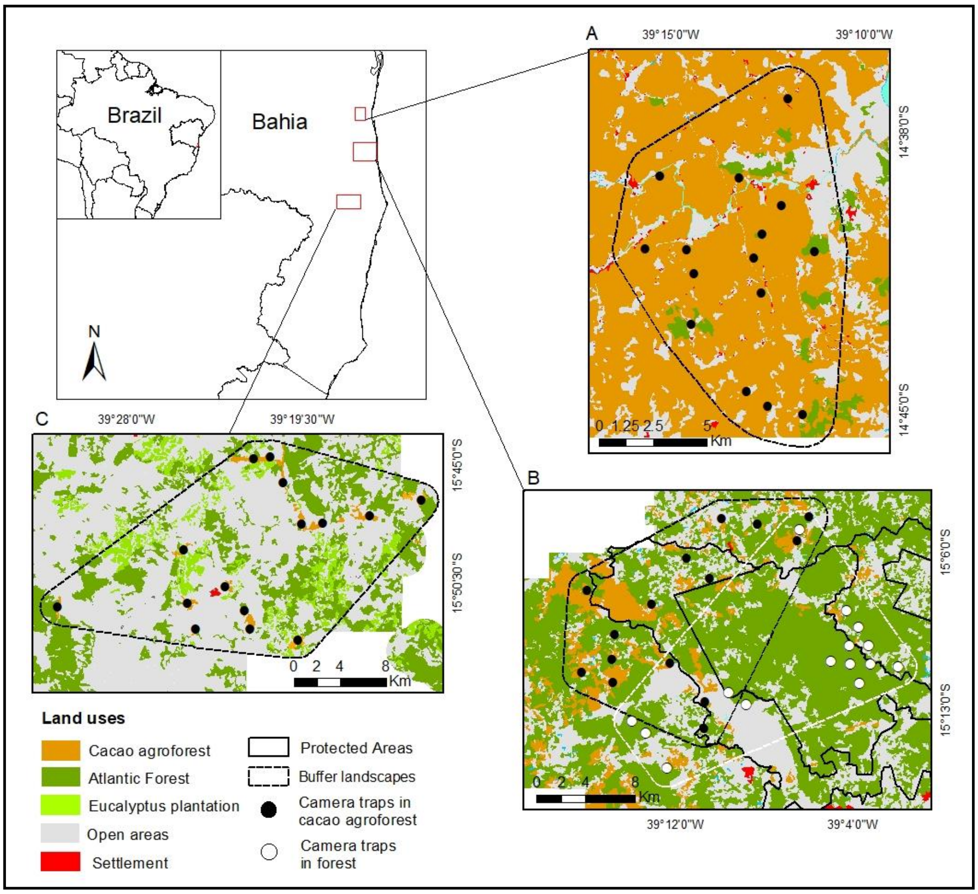

Figure 1: Study area in southern Bahia. A: the northern landscape, B: the center landscape and the control area inside and around Una Biological Reserve and Una Wildlife Refuge, and C: the southern landscape.

A northern landscape (A) was stablished in the municipality of Ilhéus and Uruçuca (80% of cacao agroforest, 5% of forest cover, and 15% of open areas); a center landscape (B) was stablished within and around Una Wildlife Refuge (16% of cacao agroforest, 54% of forest cover, and 30% of open areas; and a southern landscape (C) in the municipalities of Belmonte, Canavieiras, and Mascote (3% of cacao agroforest, 25% of forest cover, 63% of open areas, and 9% of *Eucalyptus* plantations). The control area (CA) was constituted by five forest remnants (mean 6264 ha) within Una Biological Reserve and Una Wildlife Refuge (5% of cacao agroforest, 66% of forest cover, and 29% of open areas), partly overlapping the B landscape (Figure 1).

2.2 Data collection

We carried out camera trapping surveys between June 2013 and February 2019. All sampling sites within a landscape were sampled for four months in a two-year period, 2 months in the winter (consider from April to September) and two months in the summer (consider from October to March). At each sampling site, one camera trap (Bushnell Trophy Cam) was set up at approximately 30 cm from the ground. To increase the detection rate, we used bananas and sardines as bait, following the method from Cassano et al. (2014, 2012). The cameras were programmed to take 1 photo per minute after each detection, to minimize multiple photographs of the same capture event. The camera traps were weekly visited to replace the baits and ensure proper functioning.

2.3 Data analysis:

We first used the package camtrapR (Niedballa et al. 2016) in the R software (R Core Team, 2015) to organize and extract metadata from camera traps photos. We considered an independent event of each species, when the record of the same species in the same camera was less than one hour (Massara et al. 2016).

a) Spatial occurrence:

Occupancy is defined as the proportion of area, fragments or places occupied by a species. Occupancy models allow studying the spatial patterns of species occurrence by providing unbiased estimates of species occupancy and explicitly accounting for imperfect detection (<1) (Mackenzie et al. 2002). First, we create the species detection histories for each species per survey (28 – 34 days), grouped on five occasions of 7-days of continuous sampling, matching the camera revisit schedule, a common practice in camera trap studies (Burton et al. 2015). For each occasion, the detection of the species was represented by "1" and the non-detection by "0" (Mackenzie et al. 2002). The detection history consisted of five occasions (weeks), although the final week was incomplete. To deal with this incomplete week and with the malfunction of some cameras, the sampling effort was included as an additional covariate in the detection probability.

Each mesocarnivore has different habitat adaptations in relation to forest cover and type of environment (Cassano et al. 2012; Regolin et al. 2017; Dechner et al. 2018). Thus, we also included the percentage of forest cover and agroforest cover of each agroforestry landscape and CA, as covariates that may influence in the occupancy and detection probability of native mesocarnivores, as well as the type of environment (i.e. agroforest and forest). Additionally, we included the three landscapes, CA, and season (i.e. winter and summer) as covariates.

We used a correlation matrix to determine whether pairs of covariates were highly correlated ($r \geq 0.7$). The forest cover was highly correlated to agroforest cover, as well the landscape to forest cover, agroforest cover, and environment. Thus, we excluded the agroforest cover and landscape from the subsequent analyses (Supplementary Data 1).

Occupancy models were implemented in three steps. Step 1: Because the surveys were conducted in six years and in the two main seasons (winter and summer), we first checked if our data were better explained by the "single-season occupancy model" or the "multi-season occupancy model". The first model has two parameters: the occupancy probability (Ψ) and the detection probability (p), while the second also includes the parameters local extinction (ε) and local colonization probabilities (γ) that may occur between the seasons (MacKenzie et al. 2003). Local colonization is defined as the probability of an unoccupied site during time t becoming occupied at $t+1$, while local extinction is defined as the probability of a previously occupied site during time t becoming unoccupied at $t+1$ (MacKenzie et al., 2006). In this first step we fixed Ψ as constant in the three agroforestry landscapes and the CA $\Psi(.)$, to model ε, γ, and p as function of time and seasons. We were interested to know if the ε and γ were constant among the seasons $\varepsilon(.)\ \gamma\ (.)$, or varied between them ε (t) γ (t); and if p was constant among the season $p(.)$, varied between them $p(t)$, or as function of the seasons (i.e. winter and summer) $p(sea)$. As our results were better explained by the multi-season occupancy model (excepted for *Leopardus spp.*), we continue the analysis with this dynamic modeling (Supplementary Data 2).

Step 2: We selected the top-ranked models (ΔAICc$\leq$2) from the first step. Them, in those models we modeled the Ψ as: a) constant in the agroforestry landscapes and the CA Ψ(.), b) to vary according to the type of environment (i.e. agroforest or forest) Ψ(env), and c) to vary in function of the amount of forest cover in each landscape and the CA Ψ(for). Additionally, we modeled *p* in function of the sampling effort *p*(eff) and type of environment (i.e. agroforest or forest) *p*(env). For covariates description see Supplementary Data 3.

Step 3: To understand the influence of dogs in the occupancy and detection probability on native mesocarnivores, we selected the top-ranked models (ΔAICc$\leq$2) from the second step, and we included the independent events of dogs as covariate in Ψ and *p*. We followed this three-steps procedure for each native mesocarnivores, but given the low events of *Leopardus spp*. (see Results) we could not run properly the models for this taxa. Also, the models that did not present numeric convergence of the likelihood estimate were excluded from the model selection procedure. Occupancy models were run in the program Mark (White and Burnham 1999).

b) Temporal co-occurrence:

Based on dogs dominance over mesocarnivores and the high frequency of independent events of dogs in agroforest sites (see Results), we consider dogs as a key factor that may influence on the activity patterns of native species in cacao agroforestry landscapes. Thus, we compared the diel activity pattern of each native mesocarnivores between agroforestry landscapes and the CA. For that, we used the kernel density analysis, a non-parametric method for evaluating the probability density function of a random variable (Worton 1989).

Because each agroforestry landscape was surveyed on different years and has different percentage of forest cover, we first tested whether the species differ significantly in their activity patterns within each agroforestry landscape, and if not, we pooled the data for each species in the successive analysis. The non-parametric circular Mardia-Watson-Wheeler test (Batschelet 1981), showed no difference in the activity patterns of each species among the landscapes ($p > 0.09$ in all occasions), thus we pooled the data (Supplementary Data 5). Then, to statistically compare the activity patterns of each mesocarnivore between the agroforestry landscapes and the CA we also run the Mardia-Watson-Wheeler test. This test can only be performed with ≥10 registers per species per landscape, thus we excluded *Leopardus spp*. for this analysis. This test was done with the Circular package (Agostinelli and Lund 2017) in the R software (R Core Team, 2015).

Additionally, we calculated the overlap coefficient (Δ) of each mesocarnivore between the agroforestry landscapes and the CA. This coefficient ranges from 0 (no overlap) to 1 (complete overlap), and is obtained by taking the minimum of the density functions of the two cycles being compared at each time point (Ridout and Linkie 2009). The overlap analysis mainly used two estimators, Δ1 and Δ4. The first one is recommended for smaller samples sizes (< 50 events), while the Δ4 is recommended for larger sample sizes (> 50 events) (Ridout and Linkie 2009). We used the Δ4 for *Cerdocyon thous, Procyon cancrivrorus, Nasua nasua,* and *Eira barbara,* but we used Δ1 for *Leopardus spp.* The precision of the estimators was obtained through 95% confidence intervals from 10000 bootstrap samples, and the analyses were run using the Overlap package (Niedballa et al. 2016) in the R software (R Core Team, 2015).

Results

With 6355 camera/days we recorded 2280 independents events of two exotic and 12 native carnivores (Table 1). From these species, we analyzed the data of the free-ranging dog (*Canis familiaris*), and the native mesocarnivores: crab-eating fox (*Cerdocyon thous*), crab-eating raccoon (*Procyon cancrivorus*), South American coati (*Nasua nasua*), tayra (*Eira barbara*), and because of the low events and similar ecological characteristics of margay (*Leopardus wiedii*), southern tiger cat (*Leopardus guttulus*), and no-identified small cats *Leopardus sp.* we pooled its records to one single group: small wild cats (*Leopardus spp.*). We did not analyze the data of three others mesocarnivores because of its low frequency of events (<5): the Neotropical otter (*Lontra longicaudis*), jaguarundi (*Puma yagouaroundi)* and striped hog-nosed skunk (*Conepatus semistriatus*). Even though we recorded enough events of kinkajou (*Potos flavus*) and puma (*Puma concolor*), we did not analyze its data because were almost exclusively registered in the CA. However, we present the activity patterns of both species (Supplementary Data 6).

Table 1: Independents events of exotic and native carnivores in three cacao agroforestry landscapes (A, B, C) and a control area (CA) of the Atlantic Forest of southern Bahia, Brazil.

Common name	**Species**	**Landscape**			**CA**	**Total**
		A	**B**	**C**		
	Leopardus sp.	2	5	3	3	13
Margay	*Leopardus wiedii*	0	2	4	2	8
Southern tiger cat	*Leopardus guttulus*	0	0	2	0	2
Jaguarundi	*Puma yagouaroundi*	0	1	0	0	1
Puma	*Puma concolor*	0	1	0	10	11
Crab-eating fox	*Cerdocyon thous*	61	622	248	27	958
Neotropical otter	*Lontra longicaudis*	1	0	0	0	1
Tayra	*Eira barbara*	13	57	82	422	574
Striped hog-nosed skunk	*Conepatus semistriatus*	0	0	5	0	5
South American coati	*Nasua nasua*	0	5	10	44	59
Kinkajou	*Potos flavus*	0	0	0	88	88
Crab-eating raccoon	*Procyon cancrivorus*	27	60	104	16	207
Domestic dog	*Canis familiaris*	110	108	82	21	321
Domestic cat	*Felis catus*	0	2	30	0	32

a) Spatial occurrence

The influence of dogs in the occupancy and detection probability of native mesocarnivores varied among each species (Table 2). Dogs did not influence the occupancy nor the detection probability of *Cerdocyon thous.* Its occupancy was positively influenced by the forest cover in the landscape, while its detection was influenced by the seasons (i.e. winter and summer), been significantly higher in the winter (Figure 2).

The evidence that dogs negatively influenced on the occupancy of *Procyon cancrivorus* was weak, because the model that include dogs had lower w_i value than the null model, additionally to the large confidence of interval. On the other hand, the detection probability of *P. cancrivorus* was influenced by the type of environment (i.e. agroforest and forest), been significantly higher in the agroforest. Dogs had no influence on the occupancy of *Nasua nasua*, instead it was positively influenced by the forest cover in the landscape. However, its detection probability was negatively influenced by dogs (Figure 2). There is a trend that dogs negatively influenced in the occupancy of *Eira barbara,* but is not conclusive because its large confidence interval. On the other hand, the detection probability of *E. barbara* was influenced by the type of environment (i.e. agroforest and forest), been significantly higher in the forest sites (Figure 2).

Table 2: Multi-season occupancy models of native mesocarnivores in three cacao agroforestry landscapes and a control area (large forest remnants). Ψ = occupancy, p = detection probability, ε = extinction, and γ = colonization. For covariates codes description see Supplementary Data 1. Bold rows indicate equally plausible models (ΔAICc ≤ 2).

Species	Model	AICc	Δ AICc	AICc W	k	Deviance	-2log(L)
Cerdocyon thous	**Ψ(for) ε(t) γ(t) p(sea)**	**847.168**	**0**	**0.94712**	**12**	**821.696**	**821.696**
	Ψ(for) ε(t) γ(t) p(dog)	853.373	6.2047	0.04257	12	827.901	827.901
	Ψ(dog) ε(t) γ(t) p(sea)	856.277	9.1089	0.00996	12	830.805	830.805
	Ψ(dog) ε(t) γ(t) p(dog)	862.962	15.7943	0.00035	12	837.491	837.491
Procyon cancrivorus	**Ψ(.) ε(.) γ(.) p(env)**	**583.201**	**0**	**0.49544**	**5**	**572.927**	**572.927**
	Ψ(dog) ε(.) γ(.) p(env)	**584.354**	**1.1527**	**0.27841**	**6**	**571.968**	**571.968**
	Ψ(for) ε(.) γ(.) p(env)	**584.847**	**1.6465**	**0.2175**	**6**	**572.462**	**572.462**
	Ψ(.) ε(.) γ(.) p(dog)	592.531	9.3296	0.00467	5	582.257	582.257
	Ψ(for) ε(.) γ(.) p(dog)	593.903	10.7024	0.00235	6	581.518	581.518
	Ψ(dog) ε(.) γ(.) p(dog)	594.622	11.4211	0.00164	6	582.237	582.237
Nasua nasua	**Ψ(for) ε(.) γ(.) p(dog)**	**396.078**	**0**	**0.72313**	**6**	**383.693**	**383.693**
	Ψ(env) ε(.) γ(.) p(.)	398.623	2.5443	0.20264	5	388.349	388.349
	Ψ(for) ε(.) γ(.) p(.)	400.631	4.553	0.07422	5	390.357	390.357
	Ψ(dog) ε(.) γ(.) p(dog)	418.969	22.8903	0.00001	6	406.583	406.583
	Ψ(dog) ε(.) γ(.) p(.)	421.697	25.6191	0	5	411.423	411.423
Eira barbara	**Ψ(dog) ε(.) γ(.) p(env)**	**771.932**	**0**	**0.66018**	**6**	**759.546**	**759.546**
	Ψ(env) ε(.) γ(.) p(env)	774.455	2.5234	0.18694	6	762.07	762.07
	Ψ(for) ε(.) γ(.) p(env)	774.857	2.9257	0.15288	6	762.472	762.472
	Ψ(env) ε(.) γ(.) p(dog)	800.59	28.6584	0	6	788.205	788.205
	Ψ(dog) ε(.) γ(.) p(dog)	802.846	30.9147	0	6	790.461	790.461
	Ψ(for) ε(.) γ(.) p(dog)	803.965	32.0337	0	6	791.58	791.58

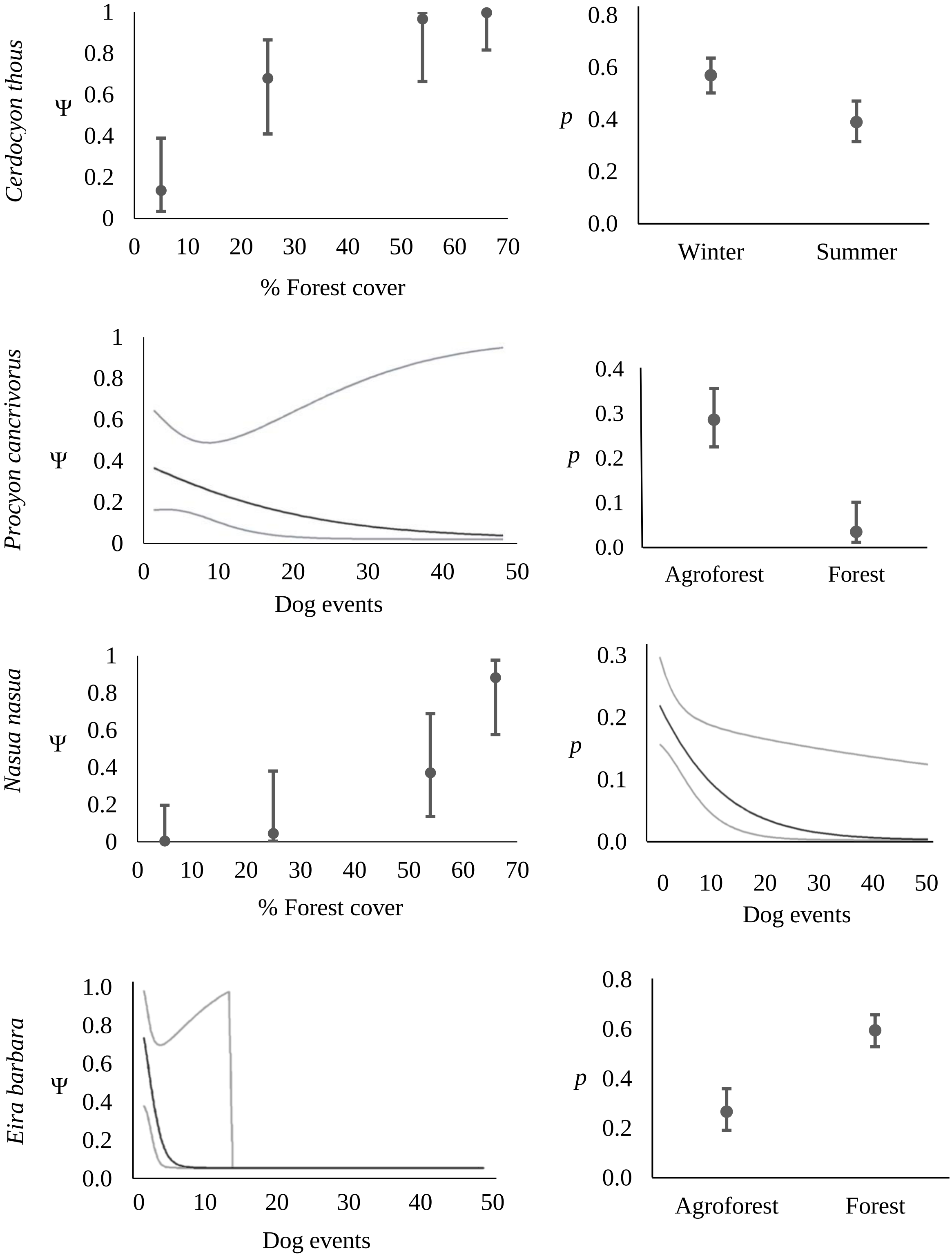

Figure 2: Mesocarnivores occupancy (Ψ) and detection probability (*p*) with its respective confidence of intervals (95%), as function of the covariates of the top-ranked model (excluding the null model for *Procyon cancrivorus*).

b) Temporal co-occurence

The activity patterns in the three agroforestry landscapes showed that *Canis familiaris* and *Eira barbara* are mainly diurnal species, while *Cerdocyon thous, Procyon cancrivorus,* and *Leopardus spp.* are mainly nocturnal. *Nasua nasua* was active during the day, dusk and night, thus it was considered as a cathemeral species. However, if we only consider its activity pattern in the CA (with low dog activity), *N. nasua* showed a strictly diurnal pattern, with a peak at mid-day. *E. barbara* showed a peak early in the morning in the agroforestry landscapes, while in the CA its activity patterns were almost stable during the day. *C. thous* is almost strictly nocturnal in the three agroforestry landscapes, while in the CA tends to be more active during day (Figure 3)

Regarding its temporal overlap between each species in the cacao agroforestry landscapes and the CA, *Leopardus spp.* presented de lowest overlap ($\Delta = 0.48$ with 95% CI = 0.38-0.61). *N. nasua* was the following with lowest overlap ($\Delta = 0.54$ with 95% CI = 0.45-0.68), and its activity pattern was different between the agroforestry landscapes and the CA (W = 17.588, *d.f.* = 2, $p < 0.001$). The third species with lowest overlap was *C. thous* ($\Delta = 0.74$ with 95% CI = 0.54-0.86) and its activity pattern was different between the agroforestry landscapes and the CA (W = 6.124, *d.f.* = 2, $p = 0.045$). The second species with highest overlap was *P. cancrivorus* ($\Delta = 0.77$ with 95% CI = 0.66-0.88), and its activity pattern was not different between the agroforestry landscapes and the CA (W = 0.494, *d.f.* = 2, $p = 0.781$). Finally, *E. barbara* presented the highest overlap ($\Delta = 0.88$ with 95% CI = 0.84-0.96) and its activity pattern was not different between the agroforestry landscapes and the CA (W = 2.192, *d.f.* = 2, $p = 0.334$) (Figure 3).

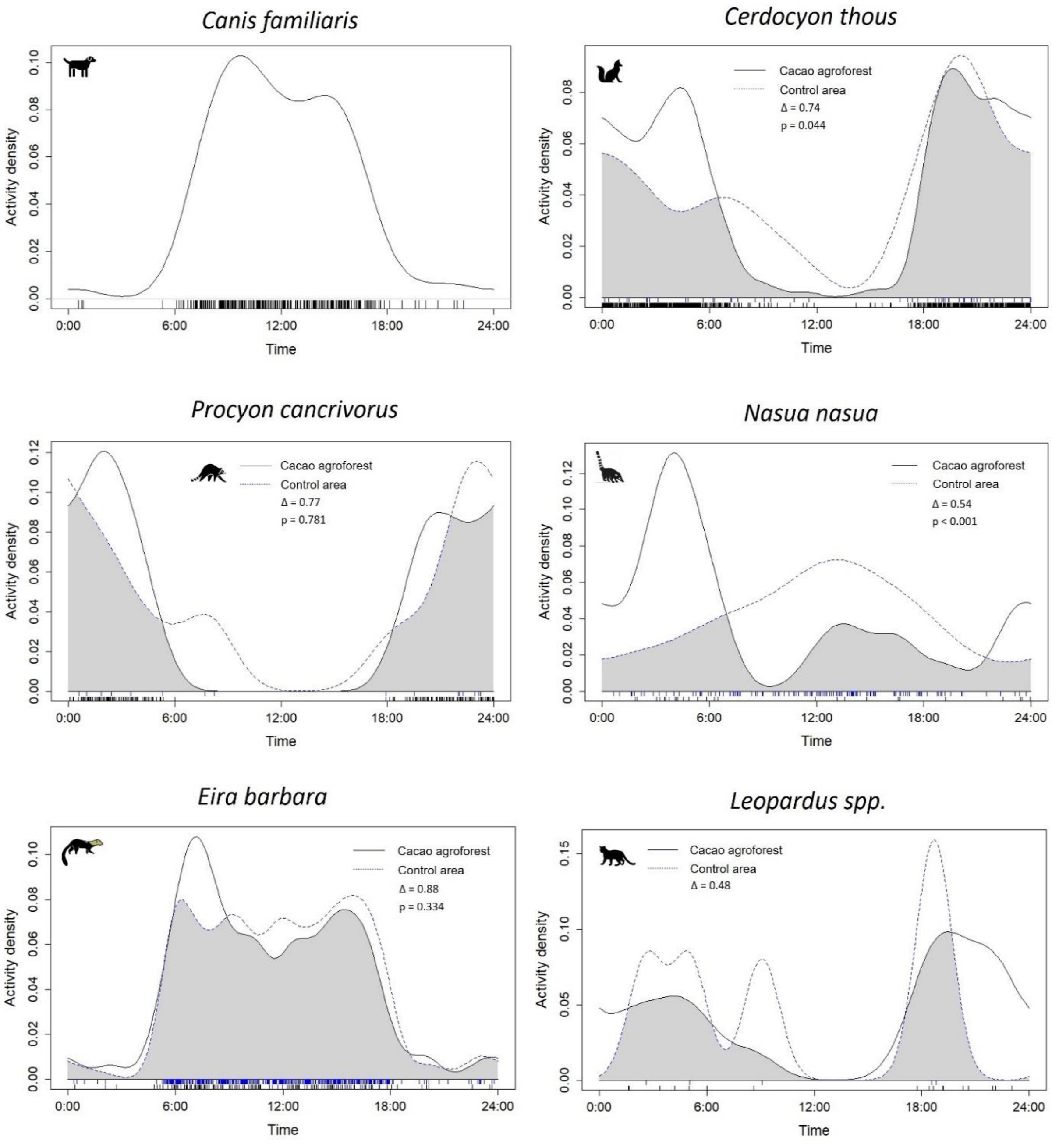

Figure 3: Overlap of diel activity patterns of each mesocarnivore in cacao agroforest (solid black lines) and a control area (dashed blue lines) in the Atlantic Forest. For dogs is represented its activity pattern from the pool data of the three agroforestry landscapes and the control area. Gray area represents the estimate overlap of both activity patterns (Δ), and *p* values show the significance. The x-axis is the time of day, and the y-axis is the Kernel activity density.

Discussion

Two of the three mechanisms that allow the co-occurrence of sympatric species in broad landscapes are the spatial and temporal segregation (Schoener 1974). However, the influence of an exotic and dominant species can alter this natural resource partitioning over native species, that avoid sites of high exotic species occupancy or shift its activity patterns (Farris et al. 2015; Zapata-Ríos and Branch 2016). This is the scenario that we are facing in cacao agroforestry landscapes of the Atlantic Forest; where dogs trend to negatively influenced the spatial or temporal patterns of most of the native mesocarnivores. For example, *Cerdocyon thous* and *Nasua nasua* avoided temporally dogs, and dogs also negatively influenced in the detection probability of *N. nasua*. Also, there is a trend that dogs negatively influenced on the occupancy of *E. barbara*, but not on its diel activity patterns. Finally, dogs records weakly influenced the occupancy of *Procyon cancrivorus*.

According to our hypothesis, dogs did not influence on the occupancy nor detection probability of *Cerdocyon thous* in the cacao agroforestry landscapes. This indicate that this species is relatively tolerant to the presence of dogs, probably due to its generalist habitat use, and omnivore diet (Michalski et al. 2006; Rocha-Mendes et al. 2010). However, *C. thous* avoid dogs temporally, reducing its diurnal activity (Bianchi et al. 2016; Massara et al. 2016), and exhibiting an almost entirely nocturnal activity in the three agroforestry landscapes. This could be as a response to avoid direct encounters with dogs, that could chase and kill this native species (Lemos et al. 2011).

Dogs negatively influenced in the detection probability of *Nasua nasua*. This means that high frequency of events of dogs in a sampling site decrease the probability that a coati would be detected in the same sampling site. Similar to *Cerdocyon thous*, *N. nasua* shifted its diurnal activity in forests to an almost nocturnal pattern in agroforests, probably because of the predation and hunting pressure that is suffering in the Atlantic Forest (Campos et al. 2007; Castilho et al. 2017), that make it shift its activity patterns to reduce encounters with dogs and its owners.

Dogs negatively influenced the occupancy of *Eira barbara*, but did not affect its detection probability nor activity pattern. As both species are diurnal and *E. barbara* did not shift significantly its activity patterns, the mechanism that is allowing the co-occurrence of both species in a landscape scale is the spatial segregation, where the native species tend to avoid sites where dogs occur. This could be attributed to its sensible response to the presence of dogs (Cassano et al. 2014), that additionally to the exploitative and interference competition, and diseases transmission that may occur between both species (Vanak et al. 2016; Doherty et al. 2017), the fear that dogs (and its owners) exert as dominant species may also negative influence on its occupancy, as have been proofed in other mesocarnivores (Clinchy et al. 2016; Suraci et al. 2016).

Dogs did not directly influence either on the occupancy or detection probability, nor the activity patterns of *Procyon cancrivorus* in the cacao agroforestry landscapes. Apparently, this species is tolerant to the presence of dogs, as has been documented in the Cerrado biome (Lacerda et al. 2009). This may be due to its generalist habitat use, and omnivore diet (Rocha-Mendes et al. 2010; Regolin et al. 2017). Also, being *P. cancrivorus* a mainly nocturnal species and dogs diurnals, both are active at different time in a diel cycle and not directly compete for resources. Although both species are omnivores, dogs are subsidized by people (Rocha-Mendes et al. 2010; dos Santos et al. 2017) and are not use for hunting *P. cancrivorus* (Castilho et al. 2017). All these statements may explain the lack of reliable dog influence on *P. cancrivorus.*

The percentage of forest cover at each agroforestry landscape positively influenced in the occupancy of the most forest-dependent species (*Nasua nasua* and *Eira barbara*), and contrary to our expectation, on *Cerdocyon thous*. Despite *C. thous* being more commonly associated to open areas and less structured habitats (Michalski et al. 2006; Regolin et al. 2017), a study in the Atlantic Forest of Argentina found that this species was more recorded in gallery forest than in shrubland and grassland (Di Bitetti et al. 2009). Based on these results, is important to maintain native forest around the agroforest to favor the occupancy of this mesocarnivore.

Thy type of environment (i.e. agroforest and forest) influenced the detection probability of two mesocarnivores. For *Procyon cancrivorus*, its detectability was significantly higher in the agroforest than in the forest, while *Eira barbara* presented the opposite pattern. This is related to the habitat preference of both species, being *P. cancrivorus* more related to less structured habitats and *E. barbara* a more forest dependent species (Regolin et al. 2017). For *Cerdocyon thous*, the seasons (i.e. winter and summer) explained better the detection probability. The winter (when its detectability was significantly higher) coincide with the mating season, and when sub-adults are dispersing to find new and suitable habitats (Faria-Corrêa et al. 2009). Therefore, in winter *C. thous* may increase its movement patterns that could influence in the higher detectability compared to the summer.

A globally research study, concluded that different types of human disturbance, as the presence of exotic species and hunting, are turning native mammals into more nocturnal species (Gaynor et al. 2018). This pattern was clearly observed on *Nasua nasua* and *Cerdocyon thous* that presented a strong shift on its activity patterns in the agroforestry landscapes, where dog events were four times higher than in the control area. In the cacao agroforestry landscapes, *N. nasua* was more active at night than in the day; while in the CA showed it natural diel activity patterns, but not the two classical peaks on the mid-morning and mid-afternoon (Rowcliffe et al. 2014; Bianchi et al. 2016; Massara et al. 2016; Azevedo et al. 2018), instead showed one peak at noon, when dog activity decrees.

In summary, our findings suggested that in agroforestry landscapes dogs are negatively influencing native mesocarnivores spatially and/or temporally. This main result should be also valid to other types of agroforestry systems (i.e. shaded-coffee and shaded-banana plantations) and agroecosystems (i.e. tree plantations and perennial croplands), where free-ranging dogs are abundant and related to human activities (Borchert et al. 2008; Dotta and Verdade 2011; Caudill et al. 2014b; Timo et al. 2014; Dechner et al. 2018). To mitigate this impact on mesocarnivores and wildlife in general, it is urgent to control domestic and free-ranging dogs populations. To achieve that, dog sterilization campaigns are needed, as well as education programs focusing on responsible ownership of dogs, to avoid workers leading their dogs into agroforest. In the Neotropics, some initiatives to control free-ranging and feral dogs have been carried out by Brazilian and Ecuadorian environmental agencies (Lacerda et al. 2009; Zapata-Ríos and Branch 2018), which also included the removal and elimination of dog populations within and around protected areas. However, these initiatives should be continuing along the years, and not only focusing in Protected Areas and its surroundings, but also in agroecosystems that has been proved to harbor several carnivores species (Ferreira et al. 2018), and act as secondary habitat and as biological corridors for several species (Faria et al. 2007; Cassano et al. 2012).

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Supplementary Data

Supplementary Data 1:

Pearson correlation coefficients for the covariates used to model the occupancy of native mesocarnivores. For = forest cover, agr = agroforest cover, env = environment, lan = landscape.

	Dog	For	Agr	Env	Lan
Dog	1	-0.18	0.18	-0.27*	-0.28*
For		1	**-0.74***	0.68*	**0.72***
Agr			1	-0.39*	**-0.85***
Env				1	**0.77***
Lan					1

*Indicates statistical significance ($p < 0.05$).
Bold numbers indicate highly correlated covariates ($r > 0.7$).

Supplementary Data 3:

Covariates use in the multi-season occupancy model.

Covariable name	Code name	Description
Dog	dog	Independents events of dogs per sampling site
Forest cover	for	Forest cover percentage in the cacao agroforestry landscapes and control area
Agroforest cover	agr	Agroforest cover percentage in the cacao agroforestry landscapes and control area
Environment	env	Type of the environment of the sampling site: cacao agroforest or forest remnants
Landscape	lan	The three cacao agroforestry landscapes and the control area
Sampling effort	eff	Total camera/days per sampling site
Season	sea	The season when the surveys were conducted: winter or summer
Time	t	The time between surveys when a species could extinct or colonize the coming survey

Supplementary Data 2:

Single-season and multi-season occupancy models of dogs and native mesocarnivores in three cacao agroforestry landscapes and a control area (large forest remnants). Ψ = occupancy, p = detection probability, ε = extinction, and γ = colonization. Ψ was fixed as constant (.) and ε, γ, and *p* as vary as a function of time (t) and seasons (sea). Bold rows indicate equally plausible models (ΔAICc ≤ 2).

Species	Model	AICc	Δ AICc	AICc W	k	Deviance	-2log(L)
Canis familiaris	**Ψ(.) ε(.) γ(.) p(t)**	**873.3877**	**0**	**0.72879**	**8**	**474.5772**	**856.7211**
	Ψ(.) ε(.) γ(.) p(.)	876.801	3.4133	0.13225	4	486.4753	868.6192
	Ψ(.) ε(.) γ(.) p(sea)	876.8852	3.4975	0.1268	5	484.4673	866.6112
	Ψ(.) ε(t) γ(t) p(t)	881.7058	8.3181	0.01139	14	469.562	851.7058
	Ψ(.) ε(t) γ(t) p(.)	888.3024	14.9147	0.00042	10	485.1305	867.2744
	Ψ(.) ε(t) γ(t) p(sea)	888.6664	15.2787	0.00035	11	483.2831	865.427
	Ψ(.) p(.)	936.8197	63.432	0	2	550.6218	932.7657
	Ψ(.) p(sea)	937.5833	64.1956	0	3	549.3308	931.4747
	Ψ(.) p(t)	939.4536	66.0659	0	6	544.9243	927.0682
Cerdocyon thous	**Ψ(.) ε(.) γ(.) p(sea)**	**875.3969**	**0**	**0.6745**	**5**	**484.0255**	**865.1229**
	Ψ(.) ε(t) γ(t) p(sea)	**877.3458**	**1.9489**	**0.25456**	**11**	**473.009**	**854.1064**
	Ψ(.) ε(.) γ(.) p(t)	880.9303	5.5334	0.04241	8	483.1663	864.2637
	Ψ(.) ε(t) γ(t) p(t)	882.2151	6.8182	0.02231	14	471.1177	852.2151
	Ψ(.) ε(t) γ(t) p(.)	885.9178	10.5209	0.0035	10	483.7924	864.8897
	Ψ(.) ε(.) γ(.) p(.)	886.4208	11.0239	0.00272	4	497.1416	878.239
	Ψ(.) p(t)	982.9554	107.559	0	6	589.4727	970.5701
	Ψ(.) p(sea)	984.1794	108.783	0	3	596.9734	978.0708
	Ψ(.) p(.)	1005.907	130.51	0	2	620.7553	1001.853
Procyon cancrivorus	**Ψ(.) ε(.) γ(.) p(t)**	**593.5586**	**0**	**0.39193**	**8**	**218.1137**	**576.8919**
	Ψ(.) ε(.) γ(.) p(.)	**593.9974**	**0.4388**	**0.31472**	**4**	**227.0373**	**585.8155**
	Ψ(.) ε(.) γ(.) p(sea)	**595.1124**	**1.5538**	**0.18022**	**5**	**226.0602**	**584.8384**
	Ψ(.) ε(t) γ(t) p(t)	596.9349	3.3763	0.07245	14	208.1566	566.9349
	Ψ(.) ε(t) γ(t) p(.)	598.7432	5.1846	0.02934	10	218.937	577.7152
	Ψ(.) ε(t) γ(t) p(sea)	600.6453	7.0867	0.01133	11	218.6276	577.4058
	Ψ(.) p(.)	621.7619	28.2033	0	2	258.9296	617.7078
	Ψ(.) p(sea)	622.8047	29.2461	0	3	257.9179	616.6961
	Ψ(.) p(t)	625.8534	32.2948	0	6	254.6898	613.4681
Nasua nasua	**Ψ(.) ε(.) γ(.) p(.)**	**426.4174**	**0**	**0.47193**	**4**	**128.4675**	**418.2356**
	Ψ(.) ε(.) γ(.) p(sea)	**427.6798**	**1.2624**	**0.25104**	**5**	**127.6377**	**417.4058**
	Ψ(.) ε(t) γ(t) p(.)	429.4253	3.0079	0.10489	10	118.6291	408.3972
	Ψ(.) p(.)	430.9068	4.4894	0.05001	2	137.0846	426.8527
	Ψ(.) ε(t) γ(t) p(sea)	431.1883	4.7709	0.04344	11	118.1807	407.9489

	Ψ(.) ε(.) γ(.) p(t)	431.5783	5.1609	0.03574	8	125.1435	414.9116
	Ψ(.) p(sea)	431.6475	5.2301	0.03453	3	135.7708	425.5389
	Ψ(.) p(t)	435.3451	8.9277	0.00544	6	133.1917	422.9598
	Ψ(.) ε(t) γ(t) p(t)	436.5404	10.123	0.00299	14	116.7722	406.5404
Eira barbara	**Ψ(.) ε(.) γ(.) p(.)**	**809.7513**	**0**	**0.66364**	**4**	**448.8418**	**801.5695**
	Ψ(.) ε(.) γ(.) p(sea)	**811.7068**	**1.9555**	**0.24963**	**5**	**448.7051**	**801.4329**
	Ψ(.) ε(t) γ(t) p(.)	814.9113	5.16	0.05029	10	441.1555	793.8832
	Ψ(.) ε(.) γ(.) p(t)	816.8596	7.1083	0.01898	8	447.4652	800.193
	Ψ(.) ε(t) γ(t) p(sea)	817.1215	7.3702	0.01665	11	441.1544	793.8821
	Ψ(.) ε(t) γ(t) p(t)	823.1724	13.4211	0.00081	14	440.4447	793.1724
	Ψ(.) p(t)	904.9901	95.2388	0	6	539.877	892.6047
	Ψ(.) p(sea)	915.721	105.97	0	3	556.8847	909.6124
	Ψ(.) p(.)	915.8439	106.093	0	2	559.0622	911.7899
Leopardus spp.	**Ψ(.) p(sea)**	**198.4968**	**0**	**0.66169**	**3**	**-91.0308**	**192.3882**
	Ψ(.) ε(.) γ(.) p(sea)	202.3194	3.8226	0.09786	5	-91.3735	192.0455
	Ψ(.) p(t)	202.7117	4.2149	0.08043	6	-93.0925	190.3264
	Ψ(.) ε(.) γ(.) p(.)	202.8354	4.3386	0.0756	4	-88.7653	194.6536
	Ψ(.) p(.)	203.0158	4.519	0.06908	2	-84.4572	198.9618
	Ψ(.) ε(.) γ(.) p(t)	206.2644	7.7676	0.01361	8	-93.8212	189.5978
	Ψ(.) ε(t) γ(t) p(.)	211.3607	12.8639	0.00106	10	-93.0863	190.3327
	Ψ(.) ε(t) γ(t) p(sea)	212.3694	13.8726	0.00064	11	-94.289	189.1299
	Ψ(.) ε(t) γ(t) p(t)	218.9338	20.437	0.00002	14	-94.4851	188.9338

Supplementary Data 4:

Multi-season occupancy models of dogs and native mesocarnivores in three cacao agroforestry landscapes and a control area (large forest remnants). Ψ = occupancy, p = detection probability, ε = extinction, and γ = colonization. For covariates codes description see Supplementary Data 1. Bold rows indicate equally plausible models (ΔAICc ≤ 2).

Species	Model	AICc	Δ AICc	AICc W	k	Deviance	-2log(L)
Canis familiaris	**Ψ(.) ε(.) γ(.) p(env)**	**863.142**	**0**	**0.5359**	**5**	**852.868**	**852.868**
	Ψ(for) ε(.) γ(.) p(env)	**864.889**	**1.7475**	**0.22368**	**6**	**852.504**	**852.504**
	Ψ(env) ε(.) γ(.) p(env)	**865.121**	**1.9794**	**0.19919**	**6**	**852.736**	**852.736**
	Ψ(env) ε(.) γ(.) p(eff)	870.399	7.2574	0.01423	6	858.014	858.014
	Ψ(.) ε(.) γ(.) p(eff)	870.734	7.5924	0.01203	5	860.46	860.46
	Ψ(for) ε(.) γ(.) p(eff)	871.567	8.4252	0.00794	6	859.182	859.182
	Ψ(.) ε(.) γ(.) p(t)	873.388	10.2459	0.00319	8	856.721	856.721
	Ψ(env) ε(.) γ(.) p(t)	873.937	10.7953	0.00243	9	855.1	855.1
	Ψ(for) ε(.) γ(.) p(t)	875.009	11.8674	0.00142	9	856.172	856.172
	Ψ(for) ε(t) γ(t) p(sea)	**847.168**	**0**	**0.86139**	**12**	**821.696**	**821.696**

Cerdocyon thous	Ψ(env) ε(t) γ(t) p(sea)	851.594	4.4263	0.0942	11	828.355	828.355
	Ψ(.) ε(t) γ(t) p(sea)	854.22	7.0515	0.02535	11	830.98	830.98
	Ψ(for) ε(t) γ(t) p(env)	855.229	8.0614	0.0153	12	829.758	829.758
	Ψ(for) ε(t) γ(t) p(eff)	858.624	11.4563	0.0028	12	833.153	833.153
	Ψ(env) ε(t) γ(t) p(env)	861.911	14.7433	0.00054	11	838.672	838.672
	Ψ(env) ε(t) γ(t) p(eff)	863.89	16.7224	0.0002	11	840.651	840.651
	Ψ(.) ε(t) γ(t) p(env)	864.469	17.3012	0.00015	11	841.23	841.23
	Ψ(.) ε(t) γ(t) p(eff)	866.212	19.044	0.00006	11	842.973	842.973
	Ψ(env) ε(.) γ(.) p(sea)	871.798	24.6298	0	5	861.524	861.524
	Ψ(for) ε(.) γ(.) p(sea)	873.333	26.1653	0	6	860.948	860.948
	Ψ(for) ε(.) γ(.) p(env)	873.978	26.8096	0	6	861.592	861.592
	Ψ(.) ε(.) γ(.) p(sea)	875.397	28.2289	0	5	865.123	865.123
	Ψ(env) ε(.) γ(.) p(env)	878.144	30.9759	0	5	867.87	867.87
	Ψ(.) ε(.) γ(.) p(env)	881.217	34.0491	0	5	870.943	870.943
	Ψ(env) ε(.) γ(.) p(eff)	882.548	35.3799	0	5	872.274	872.274
	Ψ(for) ε(.) γ(.) p(eff)	884.758	37.5902	0	6	872.373	872.373
	Ψ(.) ε(.) γ(.) p(eff)	888.478	41.3101	0	5	878.204	878.204
Procyon cancrivorus	**Ψ(.) ε(.) γ(.) p(env)**	**583.201**	**0**	**0.57389**	**5**	**572.927**	**572.927**
	Ψ(for) ε(.) γ(.) p(env)	**584.847**	**1.6465**	**0.25194**	**6**	**572.462**	**572.462**
	Ψ(env) ε(.) γ(.) p(eff)	586.606	3.4047	0.10459	6	574.22	574.22
	Ψ(.) ε(.) γ(.) p(eff)	589.419	6.2185	0.02562	5	579.145	579.145
	Ψ(for) ε(.) γ(.) p(eff)	590.909	7.7076	0.01217	6	578.523	578.523
	Ψ(env) ε(.) γ(.) p(t)	591.435	8.2339	0.00935	9	572.598	572.598
	Ψ(env) ε(.) γ(.) p(.)	591.624	8.4232	0.00851	5	581.35	581.35
	Ψ(.) ε(.) γ(.) p(t)	593.559	10.3577	0.00323	8	576.892	576.892
	Ψ(env) ε(.) γ(.) p(sea)	593.719	10.5184	0.00298	6	581.334	581.334
	Ψ(.) ε(.) γ(.) p(.)	593.997	10.7965	0.0026	4	585.816	585.816
	Ψ(for) ε(.) γ(.) p(t)	594.76	11.5593	0.00177	9	575.923	575.923
	Ψ(.) ε(.) γ(.) p(sea)	595.112	11.9115	0.00149	5	584.838	584.838
	Ψ(for) ε(.) γ(.) p(.)	595.548	12.3469	0.0012	5	585.274	585.274
	Ψ(for) ε(.) γ(.) p(sea)	596.732	13.531	0.00066	6	584.347	584.347
Nasua nasua	**Ψ(env) ε(.) γ(.) p(.)**	**398.623**	**0**	**0.55872**	**5**	**388.349**	**388.349**
	Ψ(for) ε(.) γ(.) p(.)	400.631	2.0087	0.20465	5	390.357	390.357
	Ψ(for) ε(.) γ(.) p(sea)	401.25	2.6279	0.15016	6	388.865	388.865
	Ψ(for) ε(.) γ(.) p(eff)	402.373	3.7509	0.08564	6	389.988	389.988
	Ψ(.) ε(.) γ(.) p(env)	411.665	13.0426	0.00082	5	401.391	401.391
	Ψ(.) ε(.) γ(.) p(.)	426.417	27.7949	0	4	418.236	418.236
	Ψ(.) ε(.) γ(.) p(sea)	427.68	29.0573	0	5	417.406	417.406
	Ψ(.) ε(.) γ(.) p(eff)	428.489	29.8666	0	5	418.215	418.215
Eira barbara	**Ψ(env) ε(.) γ(.) p(env)**	**774.455**	**0**	**0.54427**	**6**	**762.07**	**762.07**
	Ψ(for) ε(.) γ(.) p(env)	**774.857**	**0.4023**	**0.4451**	**6**	**762.472**	**762.472**
	Ψ(.) ε(.) γ(.) p(env)	782.328	7.8727	0.01062	5	772.054	772.054
	Ψ(env) ε(.) γ(.) p(eff)	796.699	22.2435	0.00001	6	784.313	784.313

	Ψ(envor) ε(.) γ(.) p(.)	798.612	24.1568	0	5	788.338	788.338
	Ψ(for) ε(.) γ(.) p(eff)	800.154	25.6989	0	6	787.769	787.769
	Ψ(env) ε(.) γ(.) p(sea)	800.461	26.0063	0	6	788.076	788.076
	Ψ(for) ε(.) γ(.) p(.)	801.958	27.5029	0	5	791.684	791.684
	Ψ(for) ε(.) γ(.) p(sea)	803.788	29.3327	0	6	791.402	791.402
	Ψ(.) ε(.) γ(.) p(eff)	807.437	32.9822	0	5	797.163	797.163
	Ψ(.) ε(.) γ(.) p(.)	809.751	35.2963	0	4	801.57	801.57
	Ψ(.) ε(.) γ(.) p(sea)	811.707	37.2518	0	5	801.433	801.433

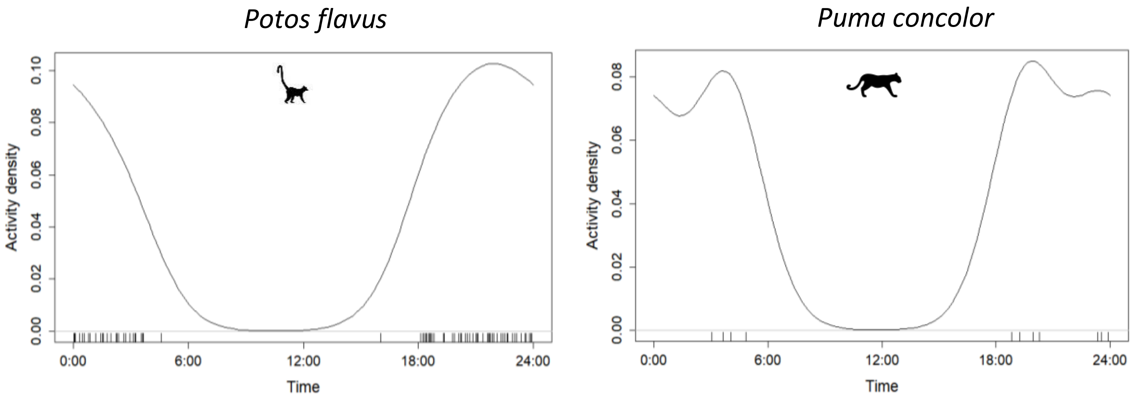
Supplementary Data 5:

Mardia-Watson-Wheeler tests among the three landscapes (A, B and C) per species. W is the static of the test, and large W values indicate greater differences between species distributions. Statistically difference for p< 0.05

Species	W	d.f	p
Canis familiaris	2.895	4	0.5755
Cerdocyon thous	4.869	4	0.0876
Eira barabara	2.415	4	0.6599
Nasua nasua	2.034	4	0.3617
Procyon cancrivorus	3.239	4	0.5186

Supplementary Data 6:

Diel activity patterns of kinkajou (*Potos flavus*) and puma (*Puma concolor*) in the control area, large forest remnants of the Atlantic Forest. The x-axis is the time of day, and the y-axis is the Kernel activity density and is not the same for both figures.



Potos flavus

Puma concolor

III. CONSIDERAÇÕES FINAIS

Os resultados mostram que em paisagens agroflorestais os cachorros rurais de vida livre estão influenciando negativamente os mesocarnívoros nativos espacialmente e/ou temporalmente. Esta conclusão deve ser válido também para outros sistemas agroflorestais (e.g. agroflorestas de café ou banana) e agroecossistemas (e.g. plantações de árvores e plantações perenes), nos quais os cachorros são abundantes e estão relacionados às atividades humanas (Borchert et al. 2008; Dotta and Verdade 2011; Caudill et al. 2014b; Timo et al. 2014; Dechner et al. 2018). Para mitigar este impacto nos mesocarnívoros e na vida selvagem em geral, é necessário controlar as populações dos cachorros rurais de vida livre e os ferais. Para tanto, campanhas de esterilização de cães são urgentes, assim como programas de educação voltados para a posse responsável de cachorros. Nos Neotrópicos, algumas iniciativas para o controle de cachorros rurais de vida livre e ferais foram realizadas por órgãos ambientais brasileiros e equatorianos (Lacerda et al. 2009; Zapata-Ríos and Branch 2018), que também incluiu a remoção e eliminação de populações de cachorros dentro de e fora de unidades de conservação. No entanto, essas iniciativas devem continuar ao longo dos anos, concentrando-se não somente em unidades de conservação e seu entorno, mas também em agroecossistemas que também abrigam várias espécies de carnívoros (Ferreira et al. 2018), e atuam como habitat secundário e corredores biológicos para diferentes espécies (Faria et al. 2007; Cassano et al. 2012).

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