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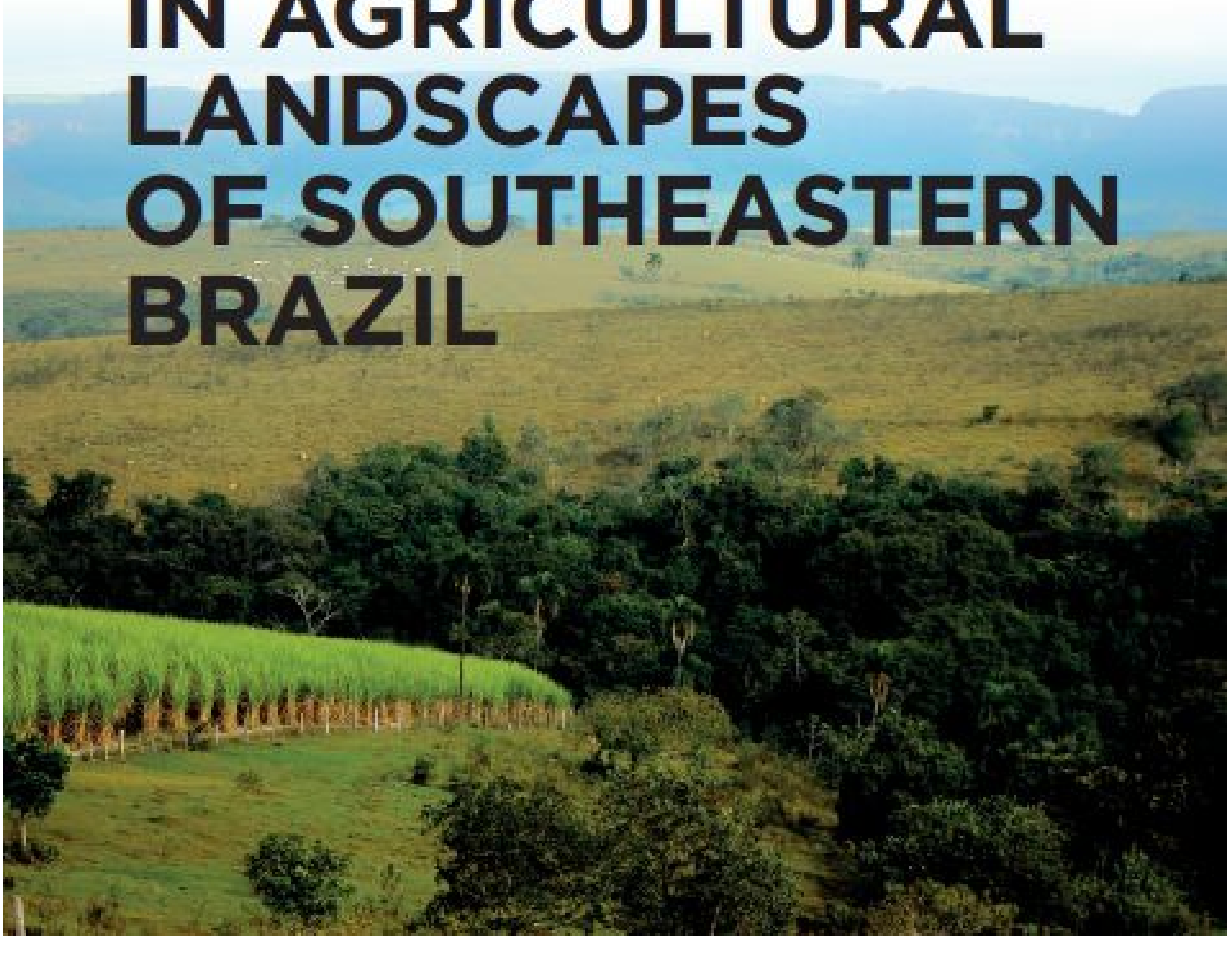
BIODIVERSITY IN AGRICULTURAL LANDSCAPES OF SOUTHEASTERN BRAZIL



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Biodiversity in Agricultural Landscapes of Southeastern Brazil

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4 Dealing with Fragmentation and Road Effects in Highly Degraded and Heterogeneous Landscapes

4.1 Introduction

Over the past 100,000 years human activities have caused continuous and severe disturbances in natural ecosystems, resulting in the extinction of many species (Enrlich & Enrlich, 1988). According to Wilson (1988), these activities continue to sharply rise, leading to the local and regional extinction of hundreds of plant and animal species, many of which are yet unknown. Issues involving endangered species and conservation of natural resources are critical, especially in tropical and developing countries. Although great effort has been made in order to reduce biodiversity loss worldwide, in many regions habitat loss and fragmentation continue to negatively influence species and key ecological processes, particularly in tropical regions (Butchart et al., 2010).

The spatial arrangement of natural remnants in the landscape is an essential issue that could favor or hamper ecological processes such as dispersion and movement of individuals to succeed in maintaining the population of other animals and plants dispersed by them. In addition, the effects of reduced vegetation cover and the spatial arrangement of vegetation remnants, urban-agroecosystems and roads in the landscape have been shown to be important elements of different ecological aspects. Roads are vital to the economic growth of a nation and permit the creation of new service opportunities, jobs and the installation of new residential and industrial areas (Wilkie et al., 2000). Therefore, we should consider that traffic, the size of opening roads and the effect of barriers created by road construction are components that should be considered at the landscapes scale.

The growth of road networks has been causing deleterious effects on wildlife since it increases the risk of mortality by vehicles (Gunson et al., 2011) and profoundly changes the amount and arrangement of natural remnants (Frair et al., 2008). Studies indicate that the unwillingness or inability of vertebrates to cross roads can lead to isolation of animal populations, with negative effects on local and regional species maintenance (Jaeger et al., 2005; Taylor & Goldingay, 2010). The reduction in abundance of a great number of species is therefore likely to be influenced by the increase of road density (Frair et al., 2008).

According to Laurance et al. (2009), when a road leads to the fragmentation process in a natural landscape, there is an inhibition of dispersal and migration of species and a growth in processes that increase the impacts on wildlife (e.g. fire, pollution, biological invasion and hunting). Consequently, fragmentation of natural areas by roads negatively affects species that: i) do not survive well in areas under the influence of edge habitats, ii) are sensitive to human presence, iii) occur in low densities, iv), tend to avoid or are unable to cross roads (Van der Ree et al., 2011). Thus, it is important to identify which animal species use roads in some way, and stimulate the development of new studies about the use and behavior of these species along the roads. This would allow stakeholders, government and other sectors of society to predict the effects of roads on wildlife and on key ecological processes, and to develop actions to mitigate the impacts.

The present text overviews the advances in studies of cover and spatial structure of habitats and road effects on different groups within highly fragmented and heterogeneous landscapes. Using a spatial ecology approach, we also propose a potential research agenda for these fields, in order to stimulate new studies focusing on key issues to maintain and recovery biodiversity within agroecosystem-dominated landscapes. We focus on these issues with examples of studies conducted mostly in the Neotropical region due to its importance in terms of biodiversity and presence of agroecosystems and also because of the growing number of studies carried out in this region, permitting the detection of gaps in knowledge to which research efforts should be directed. We start by discussing general aspects of agroecosystems, roads and edge effects. Then we outline studies using genetics and stable isotope analysis for functional groups, which are useful tools for understanding landscape effects. Finally, we provide an overview of road and edge effects on two example animal groups, bats and ants, and we finish with highlights on landscape ecology and agroecosystems.

4.1.1 Urban-agroecosystem Gradients and Biodiversity

The urbanization process leads to large variation in the composition of natural communities and, therefore, in ecological dynamics. While many species are lost due to the suppression of their habitats, the development of the anthropic landscape also brings forth new micro-climatic spaces favorable to the settlement and development of species that cannot be found in other ecosystems in the surrounding area. This process may be seen as positive when considering the possibility of these species settling in the area, or negative, as many native species may lose their capacity to develop in this new environment, permitting the entry of exotic, more competitive, groups.

In some cases, even when exotic species are excluded from measurements of biodiversity the sub- and peri-urban systems still have more species than the environment that existed before the urbanization process (McKinney, 2008). This may be due to many reasons, such as the unavoidable creation of new micro-habitats that may harbor new animal and plant groups and the intentional introduction of species, as is the case of cultivated areas in city outskirts, which can increase the amount of resources available for other species. These peri-urban environments are the natural entryways and outlets of the material and energy flow between the predominantly human ecosystems formed by cities and the rural and natural ecosystems.

Moreover, the variation that exists in these sites cannot be viewed as a linear gradient between the urban and rural areas. Environmental and ecological conditions depend not on a point located along a straight line, but on characteristics that are also determined by the adjacent points (Ramalho & Hobbs, 2012). Thus, an area with high biodiversity may act both as a shelter for many species and as a source of propagules into neighboring sites, increasing the complexity and heterogeneity of these gradients.

There are many ways to achieve high biodiversity in urban areas, and one method that has been shown to be highly efficient in many environmental, social and economic aspects is the development of more sustainable landscapes though agroecosystems, particularly in urban-rural gradients (Alvey, 2006; Grimm et al., 2008). This combined cultivation technique that often takes place in peripheral environments may ensure not only better soil quality, but also the possibility to maintain the ecosystem services found in the area, such as water source preservation, pollination services and greater carbon sequestration (Alberti et al., 2003).

In addition, the development of agroecosystems permits a complete association between environmental preservation, improvement of urban environments, and increases in the income of the less favored population. All this makes Agroecosystems one of the main agents of biodiversity conservation projects in cities. No less important is the approximation between society and nature, which has been one of the big key points in the success of urban ecology-related actions, as decision making by public agencies is difficult in places without a good relationship between humans and their ecosystem.

4.1.2 Road Ecology, Landscape Structure and Biodiversity

Together with the urbanization process comes the transportation system, which is an increasing issue to biodiversity - although important to humans, roads promote several negative effects to species and ecological processes. One of the main effects of roads and highways is the interruption of the dispersal of animals and plants, breaking connectivity and leading to a change in habitat quality due to fragmentation. In turn, disturbance of between-fragment movement interferes with between-population migrations and, consequently, with the populations' structure and viability (Forman & Alexander, 1998), making the impacts of anthropic interventions on species more evident. The isolation generated by fragmentation also decreases the populations' genetic diversity, interfering with the adaptive plasticity of species in face of environmental changes that take place through time (Frankham et al., 2002).

The local and global variation generated by these interventions may influence evolutionary processes, and responses to these changes require high genetic variability to ensure a species' reproductive success and ongoing existence (Sork & Smouse, 2006). High levels of genetic variation are maintained by a constant flow of individuals between populations; however, the construction and presence of roads and the lack of ecological corridors in natural environments interrupt these flows, and species' spatial patterns come to be defined by habitat removal and by the subdivision of continuous areas (Miller et al., 1996; Forman & Deblinger, 2000).

The flow of individuals between populations is not constant for all the species that exist in a given fragment – for instance, Hawbaker et al. (2006) have shown that some species avoid crossing roads, whereas other species cross them. However, the individuals that take their chances in crossing the roads end up generating high levels of roadkill along these pathways (Forman & Alexander, 1998). These individuals are usually reproductive-age adults, and their mortality contributes to the fast decreases in population size of each of the species found in these fragments (Ashley & Robinson, 1996).

Animal roadkill is the most evident direct cause of mortality of wild vertebrates that is related to fragmentation by roads (Forman & Alexander, 1998). The yearly roadkill rates estimated by Forman (1995) and by Van der Zande et al. (1980) are disturbing: 159 thousand mammals and 653 thousand birds in the Netherlands; seven million birds in Bulgaria; five million frogs and reptiles in Australia and one million vertebrates in the United States. Reports of roadkill of vertebrate species are scarce in many areas; this situation is even more worrisome when considering the wide road and highway system and the country's biodiversity.

These impacts become even greater when considering that roads encourage the building of houses (Hawbaker et al., 2006) which, in turn, encourage the construction of new roads, leading to a vicious circle. It is also important to highlight that the species' responses to change take time, increasing the conflict due to the high speed with which construction-related changes occur in the environment. Another factor to be considered is the increase in traffic volume, which may lead to avoidance effects. Thus, maintenance of habitat quality demands the reduction of construction-related impacts; however, in order to mitigate the barrier effect produced, it is fundamental to assess the impacts and to identify the species most affected. Estimating the quantity of roadkill of small vertebrates is hard, especially due to difficulties in identifying the affected animals. Therefore, despite being hard to mitigate this impact, actions are urgently needed in order to reduce the negative effects of roads to biodiversity maintenance.

4.1.3 Overview of Edge Effects in Fragmented Landscapes

In addition to the direct effects of roads and other anthropic land uses on animal mortality and connectivity, it is important to consider how these land uses affect the structure and function of the adjacent natural ecosystems. These edge effects[1] may be defined as detectable differences between forest edge and the ecosystem on either side of the edge with regard to the ecosystem's structure, composition and function (Harper et al., 2005). Edge effects may be described by two parameters, the depth or distance of edge influence (also known as "edge width") and the magnitude of edge influence (Harper et al., 2005). Depth of edge influence is a measure of how far into the fragment statistically significant edge effects may be detected, whereas magnitude of edge influence is a measure of how different the edge and interior environments are in relation to each other. Edges may be characterized by different combinations of high and low depth and magnitude of edge influence (Harper et al., 2005). A variable may be influenced positively or negatively by the edge (Ries et al., 2004), and non-monotonic responses may also be observed (e.g. Dodonov et al., 2013).

Edge effects often result from changes in microclimate, such as increased light and wind incidence, which may result in greater temperature, lower moisture and increased wind throw at the edges (Saunders, 1991; Laurance & Curran, 2008). These changes in microclimate and the resultant modifications in vegetation structure and composition may, in turn, affect the resident fauna (Meyer et al., 2009) and vegetation (Cadenasso & Picket, 2000). In addition, as the edge vegetation regenerates or undergoes changes in structure and/or composition, the extent of microclimatic edge effects will also change (Didham & Lawton, 1999).

It is important, however, to note that edge effects are not caused solely by microclimatic changes, nor are they restricted to closed forests. For example, edge-related changes in vegetation composition have been observed at road edges of grasslands in Australia and Africa (Cilliers et al., 2008) and at different types of edges in the Brazilian savanna (Dodonov et al., 2013). In these areas, edge vegetation was largely dominated by exotic species, possibly because the establishment of these species was favored by disturbances or changes in soil chemistry. Another example of edge effects resulting from linear openings was observed by Smit & Asner (2012) in an African savanna, where there was a greater abundance of woody plants close

1 Some authors (e.g. Harper et al., 2005) recommend to use the term "edge influence" instead of "edge effects" for historical reasons: whereas the term "edge effects" was initially used to describe the increase in abundance of game species at forest edges (Leopold, 1933, cited by Harper et al., 2005), "edge influence" is more general and refers to all edge-related modifications in the ecosystem. However, we preferred to use "edge effects" in this chapter because this term is more commonly found in the literature (e.g. Fahrig, 2003; Ries et al., 2004).

to firebreak roads. This edge effect could have resulted from increased water run-off from the roads, thus increasing soil moisture at the edges (Smit & Asner, 2012).

These examples show the complexity of edge effects in different environments, and a number of factors have to be considered in order to successfully account for edge effects in a landscape study. The extent of edge effects may be affected by factors such as vegetation type (Delgado et al., 2007), adjacent land use (Pohlman et al., 2007), fragment size (Mascarúa-Lopez et al., 2006) and edge age (Chabrerie et al., 2013). Areas located close to more than one edge may also suffer greater edge effects than areas close to a single edge (Fletcher, 2005). In addition, it is important to note that different variables respond to edges in different ways, and there may be marked differences in the depth and magnitude of edge influence observed for variables such as seedling mortality and species diversity (Harper et al., 2005). Finally, different statistical analyses may give different results for the same data set (Harper & Macdonald, 2011). Therefore, detailed studies are needed before the extension and characteristics of edge effects in a given region may be predicted.

4.1.4 Understanding Landscape Effects on Genetic Diversity and Gene Flow

On a larger scale, roads and other sources of habitat fragmentation may lead to important changes in the genetics of different populations. A fragmented landscape is characterized essentially by the reduction of favorable habitats and by changes in its spatial configuration (Fahrig, 2003). Habitat loss and fragmentation may affect species richness (Martensen et al., 2012) and interaction processes such as seed dispersal, as well as decrease in population abundance (Fahrig, 2003; Uriarte et al., 2011). These are also considered the main factors that lead to decreases in the genetic variability of different species (Lowe et al., 2005; Aguilar et al., 2008), mostly due to a drastic decrease in population size (e.g. Andersen et al., 2004). In addition, in populations where size has been reduced there is a greater probability of mating between related individuals which, combined with genetic drift, also decreases genetic viability (e.g. Breed et al., 2012). Low genetic variability may in turn decrease the species' adaptive potential in face of environmental changes, which may lead to extinction (Hoffmann & Willi, 2008).

Movement and dispersal capacity of individuals is also affected by landscape fragmentation (Awade & Metzger, 2008). The movement of individuals between populations maintains species' connectivity; in plants, this connectivity is preserved by pollen and seed dispersal (Sork & Smouse, 2006). Animal and plant dispersal depends on natural history and landscape characteristics, such as matrix type (Umetsu et al., 2008), and distance between favorable habitats (Boscolo & Metzger, 2011). For example, population isolation and matrix resistance may limit the dispersal of individuals between populations, reducing genetic flow and consequently population connectivity (e.g. Lange et al., 2012). This may lead to a disturbance in the migration-drift balance, as the alleles lost due to genetic drift will not be replaced by the gene flow (Hamrick, 2004).

Even though the population genetics theory predicts the effects of habitat loss and fragmentation in reducing genetic variability (Young et al., 1996), many studies do not support these predictions (e.g. Collevatti et al., 2001; Winkler et al., 2011). This may be due mostly to the recent fragmentation of the area, considering the life cycle of the species studied (Collevatti et al., 2001; see also Lowe et al., 2005 for a review). In addition, the non-detection of the effects of habitat loss and fragmentation of genetic variability and gene flow may be due to the sampling designs often used in studies of this kind, as they do not compare different landscapes and often do not explicitly test the influence of landscape characteristics on the genetic structure of populations (e.g. Chambers & Garant, 2010 and see Storfer et al., 2010 for review).

Alternatively, there may be a reduction of genetic variability due to intense population reduction, as in the case of large mammals (Roelke et al., 1993). Combining studies to understand landscape effects on genetic variability, persistence of species and ecological processes is something needed in conservation biology. We live in a defaunated world (Dirzo & Galetti, 2013) and the use of genetics and other tools, such as the one we present below, may help provide guidelines for maintaining ecological processes.

4.2 Case Studies

4.2.1 Stable Isotope Analysis and Functional Diversity: Potential Tools for Subsidize Mammal Conservation Strategies in a Landscape Perspective

Mammals are one of the most vulnerable groups to threats such as habitat loss and fragmentation (Chiarello, 1999; Galetti et al., 2009; Dotta & Verdade, 2011), hunting pressure (Cullen et al., 2000; Peres, 2000) and roadkilling (Forman & Alexander, 1998; Scoss et al., 2004; Huijser et al., 2013). Medium- and large-sized mammals are key organisms for the structuring of biological communities (Dotta & Verdade, 2007), and are involved in fundamental ecological processes, such as herbivory, predation (Cuarón, 2000) and seed dispersal (Fragoso & Huffmann, 2000; Galetti et al., 2001). Therefore, they play a unique role in the maintenance, regeneration and dynamics of tropical forests (Jorge et al., 2013), even in modified and defaunated landscapes (Galetti et al., 2013).

In studies linking species, ecosystem fragmentation, and conservation, landscape ecology has been providing new study models (Metzger, 2001). Several studies indicate that mammals use the landscape matrix (Dotta & Verdade, 2007; Ferraz et al., 2010; Ferraz et al., 2012; Magioli et al., 2014a; Magioli et al., 2014b, Chapter 11 of this book), which is currently a dominant component in most Atlantic Forest domains, occupying approximately 84% of the biome original extension (Ribeiro et al., 2009).

Thus, in order to understand ecological processes, it is important to consider these modified habitats, since studies that included the effect of matrices in heterogeneous landscapes increased the explanatory power of ecological models (Umetsu et al., 2008; Prevedello & Vieira, 2010; Watling et al., 2011).

Several studies in landscape ecology have taken into account the influence of the landscape matrices over wildlife communities (Baum et al., 2004; Murphy & Lovett-Doust, 2004; Antongiovanni & Metzger, 2005; Dotta & Verdade, 2007; Umetsu et al., 2008; Pardini et al., 2009). However, knowledge on the effects of landscape structure on medium- and large-sized mammal species is scarce (Grelle, 2010; Prevedello & Vieira, 2010), raising some questions: 1) how do mammal species use these modified habitats? 2) which food resources are available and consumed? and 3) what is the contribution of persistent species for ecosystem functioning? Seeking new ecological approaches to answer these questions requires the combination of studies in landscape ecology. Two methods present potential for the acquisition of new information on modified landscapes: stable isotope analysis and functional diversity measurements.

Stable isotope analysis (SIA) generates information on trophic ecology, feeding patterns, origin of food resources, and has been used with considerable success in mammal studies (Kelly, 2000; Crawford et al., 2008; Ben-David & Flaherty, 2012). Carbon ($^{13}C/^{12}C$) and nitrogen ($^{15}N/^{14}N$) stable isotopes are used to assess animal diet and resource use, which also provides insight into trophic chain processes (De Niro & Epstein, 1978; Boecklen et al., 2011), and represents a useful tool for obtaining information on highly modified landscapes (Magioli et al., 2014b). SIA requires a tissue sample from the species of interest, with muscles and bone collagen being the most often used (Ben-David & Flaherty, 2012). Collecting these tissues requires the capture and sacrifice of the specimens, a strategy that is not particularly easy or ideal for studying rare or threatened species. Alternatively, the analysis of hair or feces offers a non-invasive method (Magioli et al., 2014b).

Thus, SIA can provide important information on trophic ecology, being complementary to diet studies and can help infer changes in habitat and resource use over both short and long time periods, which is of great value for species conservation (Kelly, 2000; Sponheimer et al., 2003; Codron et al., 2005; Crawford et al., 2008; Magioli et al., 2014b). However, the proper use of this tool requires i) a carefully elaborated sampling design, ii) questions and objectives that SIA may be able to answer, and iii) comparison of isotopic results with ecological and behavioral data of the studied species to avoid misinterpretations.

Functional diversity measures are presented as tools that may offer insights on the relationships between species and ecosystem functioning (Cadotte et al., 2011), and are being applied in various ecology fields and with different taxonomic groups (Cianciaruso et al., 2009). Functional diversity assesses the amount of interspecific variation in functional characteristics of a community, treating each species as unique (Poos et al., 2009).

There are several distance-based functional diversity measures (Mouchet et al., 2010), which can use species presence/absence data, such as the functional diversity index (FD) (Petchey & Gaston, 2002, 2006) and functional richness (FRic) (Villéger et al., 2008). Also, metric on abundance data can be used, such as Rao's quadratic entropy (Q) (Rao, 1982), functional evenness (FEve), and functional divergence (FDiv) (Villéger et al., 2008). When related to landscape characteristics, ecosystem processes and other metrics, each offer distinct insights on the relationship between species and ecosystem functioning. One of the most important things to consider when calculating functional diversity measures is to select an ecologically meaningful set of species traits. These traits need to be directly linked to the questions and objectives of the study (Cianciaruso et al., 2009), and preferably uncorrelated.

There is a large amount of information on the ecology and behavior of terrestrial vertebrates, particularly for mammals, which is widely used in conservation planning strategies (Jenkins et al., 2013). Over the last decade, functional diversity has been used to address several issues in mammal studies, providing insights for species conservation. Specifically, studies address the loss of functional diversity due to the intensification of land use by human activity, and the identification of thresholds between functional diversity and landscape metrics (Stevens et al., 2003; Blackburn et al., 2005; Flynn et al., 2009; Chillo & Ajeda, 2012; Magioli et al., 2015).

4.2.2 Anthropogenic Effects on Bats

Bats are a very large order of mammals in terms of diversity and abundance. They provide crucial environmental services such as pest control in agroforest ecosystems (Willians-Guillén et al., 2008) and function in ecological succession (Lobova et al., 2009). Therefore it is important to quantify the various ecological roles of bats and assess how intense human ecosystem modification affects these services. For example, bats play an important role in seed dispersal (Lobova et al., 2009), and contribute to the landscape-scale habitat regeneration of native species in agroecosystems.

Despite their abundance, bat populations are in decline in many areas (Altringham, 2011) with almost 25% of populations threatened by extinction (Mickleburgh et al., 2002). With advances in technology, new studies are able to monitor bat populations with higher accuracy (O´Mara et al., 2014), providing evidence for this population decline over decades (Betke et al., 2008).

Much has been done to understand the impact of human activities on bats, mainly in Poland and the USA. For example, great efforts have been made to minimize the effects of white nose syndrome in temperate regions (Blehert et al., 2009), which has killed thousands of bats over a decade. In the following section, we discuss two aspects of anthropogenic impacts: the presence of roads and a disease that affects both bats and humans.

4.2.3 Roads and Bats in Heterogeneous Landscapes

As optimal foragers (Pyke, 1984), bats always look for minimum cost paths in their daily routes (in migrating species this idea is more complex, see Fleming et al., 2008). Therefore, bats commonly use linear landscape features, such as open trails inside fragments, rivers and roads. Russel et al. (2009), investigated these aspects, demonstrating that bats cross roads in low flights (< 2 m high), with an annual mortality of 5%. Several papers (Shaub & Siemers, 2008; Siemers & Shaub, 2010, Berthinussen & Altringham, 2012a,b) showed an intense negative impact of roads on bat populations, but these effects remain unknown in the Neotropics, where few studies investigate this issue (Bernard et al., 2012).

Bats are common and abundant near cities and agroecosystems, being able to feed and roost in these environments (Bredt et al., 2012). Bats are also a known vector for many types of viruses (Calisher et al., 2006; Sabino-Santos Jr et al., 2015). An important issue affecting both bat and human health is rabies, a viral disease. In very altered ecosystems, such as those arising from deforestation, virus outbreaks can become more frequent due to increased contact of vampire bats with humans (Schneider et al., 2001) or alteration of viral rates on bat populations with unknown causes. Viral rates vary between different regions (Ruppretch et al., 2002) but in general the main rabies vectors are dogs and bats (Paéz et al., 2003). Several species of bats are well adapted to live in cities and houses (see Bredt et al., 2012), which increases contact risk to humans and also to herbivores in pastures (Gomes et al., 2010), such as cattle and horses.

Although instances of rabies are less common than other diseases, survival is rare once symptoms occur (Willoughby Jr et al., 2005). Therefore, strategies on how to prevent rabies are needed. This issue should be addressed mainly in cities with nearby extensive natural environments, such as in the "arc of deforestation" around the Amazon forest. In these areas, rapid infra-structural changes due to increases in agricultural crops and cattle fields can cause the rapid, opportunistic population growth of vampire bats.

In general, responses to fragmentation are still unclear for many bat species in the Neotropics (see Cunto & Bernard, 2012; but see also Ripperberger et al., 2015) and this remains especially true for agroecosystems, where more studies are still needed (Faria et al., 2007; Heer et al., 2015; Muylaert, 2015) in order to elucidate effective management strategies (see Chapter 8, this book). Moreover, isotopic analysis studies and functional diversity approaches could help determine how bats respond to heterogeneous landscape features.

4.2.4 Ants as Bioindicators of Fragmentation and Edge Effect in Neotropical Environments

Ants are among the most diverse organisms on the planet: they can inhabit any terrestrial environment, with the exception of the polar caps, and, due to their high richness and abundance, tend to be a dominant group in many habitats (Folgarait, 1988). Extreme ant biodiversity is observed in the Neotropics – for example, in the Amazon rainforest ants may represent more than 25% of the faunal biomass and over 40 species may be found in a single tree (Schultz, 2000; Wilson, 1987). This high diversity demonstrates the importance ants have in ecological processes, acting as disperses of seeds, decomposers, and agents of soil cycling and aeration (Lobry de Bruyn, 1999).

Due to their importance to ecosystems, high diversity, and the facility of sampling and identification, ants are considered good indicators of the impact of human activities on the environment (Majer, 1983; Andersen, 1993; Andersen & Majer, 2004). In addition, ant assemblage composition reflects trends in the composition of others groups more so than plants, terrestrial invertebrates and birds (Majer et al., 2007). Ant species can be grouped in functional groups (based on taxonomic, morphological, trophic and nesting behaviors) in order to reduce ecological complexity, enabling comparative analyses among environments that have few or no species in common (Philpott et al., 2010).

Compared to other regions and organism groups, there are few studies on the effect of fragmentation on ant assemblages in Neotropical natural habitats (Leal et al., 2012). Fragmentation characteristics, such as fragment size and isolation, are considered important factors driving the richness of ant assembly. On the contrary, the main factors affecting the richness of ant assemblages are the structural characteristics of vegetation and landscape, such as tree density and richness, as well as landscape heterogeneity (Carvalho & Vasconcelos, 1999; Pacheco & Vasconcelos, 2012; Ribas & Schoereder, 2007; Santos et al., 2006).

This relationship of ants with vegetation structure shows a relevant indirect effect of fragmentation on the assembly of ants. The hyperproliferation of pioneer tree species, as consequence of fragmentation, simplifies the forest structure, consequently reducing the heterogeneity and diversity of resources used by ants (nesting sites and food) (Leal et al., 2012). Studies that partitioned ant assemblages into functional groups show direct effects caused by fragmentation: functional groups such as army ants, specialist predators and cryptic species had their richness reduced when the fragment area was reduced (Leal et al., 2012; Petters et al., 2011). On the contrary, fragmentation usually benefits generalists, dominant and opportunist species of the Myrmicinae and Dolichoderinae subfamilies, as well as the leaf-cutting Attinae tribe (Dohm et al., 2011).

Edge effects influence ant assemblages in ways similar to fragmentation, reducing the diversity of some functional groups while benefiting generalist and opportunist species (Carvalho & Vasconcelos, 1999; Wirth et al., 2007; Dohm et al., 2011). Areas under edge influence usually have exotic species of ants (which generally belong to generalists, opportunistic and dominants functional groups), that directly compete with and reduce the diversity of native ants and others arthropods (Suarez et al., 1998; Holway, 2005). Some of these exotic species are among the most important invasive alien species (Lowe et al., 2000). These species often have a close relationship with human activity, which can cause accidental dispersal and offers a resource-rich environment free of natural predators, such as in urban areas, roads and powerlines (Lodge, 1993; Stiles & Jones, 1998; Suarez et al., 1998).

Therefore, studies that focus on the assembly of ants in fragmented environments are essential for Neotropical forest conservation on three different levels. First, the total diversity of ants could provide a good diagnostic measure regarding the vegetation structure and heterogeneity of the landscape. Second, by looking at functional groups of ants it is possible to see direct effects of fragmentation on ant assemblage, as well as the consequences of these changing patterns of ant assembly on ecological processes. Third, such studies could identify and delineate strategies to reduce the consequences of biological invasions caused by species of exotic ants.

4.3 Where Should We Go with Landscape Ecology within Agroecosystems?

The demand for information on how to manage landscapes in order to archive a better balance between human needs/activity and biodiversity conservation is by far higher than the information available to date. Government agencies, landowners, NGOs and a variety of industries ask for more precise information to be used in conservation planning, and prioritizing areas for conservation and restoration. But we still know very little about how habitat cover, configuration, matrix, edges and roads influences different species, functional groups and ecological processes. These are gaps that need to be urgently addressed in order to better understand the importance of landscape structure and the effects of roads on biodiversity, particularly within agriculture-dominated ecosystems:

- More than a habitat-matrix dichotomy: first we need to improve our concept of habitat, because many species use a variety of landscape features with different intensities and for different ecological functions (feeding, shelter, breeding, dispersal, escape from fire or pests). Thus, we need to avoid the idea that species interact with landscape features as habitat/matrix; however, information about how species uses natural and modified land cover types is scarce. Developing new research to effectively deal with the landscape as a complex mosaic is a great avenue for researchers during the next decade.
- Matrix types matter: we consider that it is important to broaden our vision of habitat types when classifying landscape elements, especially in diversified

landscapes such as agroecosystems. Within agroecosystems-dominated landscapes it is possible to have different types of matrices. This heterogeneity can be of good use for wildlife and also for economy, in terms of local economy and the appropriate use of soil, We need to consider how different matrices influence species persistence and movement at the landscape scale, as well as the effects of “matrix depth” (or matrix extent) on biodiversity maintenance. In this sense, we need to sample gradients of distance within the matrix in relation to habitats of reference within landscapes.

– Variation in edge effects: it is also important to consider both the vegetation type and the surrounding matrix when modelling edge effects in fragmented landscapes. Considering habitat and matrix as homogeneous, without accounting for differences in their structure, composition and function is likely to lead to an oversimplification of the proposed models.
– From patch-based to landscape-based studies: although landscape ecology advanced in different important directions during the last two decades, we need to move on toward a more landscape-based sampling design instead of patch-based surveys. Many of the studies with a landscape ecology approach still focus on patches as sampling units, often surveying only one patch or a particular type of habitat (e.g. forest). To effectively understand the full influence of landscape structure on biodiversity (see part II) we need to use landscapes as sampling unities and sample a variety of habitats and matrices within each landscape.
– Road ecology - beyond roadkill: road ecology studies became more common, but many of them still focus mainly on road kill-based research. There are plenty of ecological processes, for a great variety of taxa and regions, which need to be investigated in the light of road ecology. For example, what is the influence of road structure on facilitating or impeding road cross? What are the effects of roads on ecological processes such as frugivory, herbivory, hydrological regime, landscape regeneration etc?
– Movement ecology as a research agenda: movement is a key aspect of ecological processes because it is central to determine gene flow, dispersion, landscape regeneration etc. Understanding how landscape influences spatial-temporal movements at different scales will allow us to better define conservation and restoration priorities and strategies, especially for species or ecological processes that are more influenced by landscape spatial heterogeneity and habitat degradation. More than just knowing the home range of species, we need to effectively monitor and understand the movement patterns of keystone species, and consequently to better comprehend ecological processes that are more influenced by movements of different organisms.

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