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*Chapter 8*

# CARNIVOROUS MAMMALS IN A MOSAIC LANDSCAPE IN SOUTHEASTERN BRAZIL: IS IT POSSIBLE TO KEEP THEM IN AN AGRO-SILVICULTURAL LANDSCAPE?

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## INTRODUCTION: HABITAT FRAGMENTATION AND ITS EFFECTS ON BIODIVERSITY

Habitat fragmentation can be defined as a process where continuous areas of natural habitats are broken into small patches separated by other habitats different from the original ones (Wilcove *et al.* 1986; Andrén 1994). Today, habitat fragmentation is a common issue to almost every ecosystem in the world, since anthropogenic land uses transformed initially continuous habitats into mosaic landscapes represented by isolated native patches surrounded by man-altered environments (Nagendra *et al*. 2003).

Anthropogenic matrices usually act as selective filters to species movements among native patches in the landscape (Gascón *et al.* 1999), and therefore, the persistence of animal and plant populations in fragmented habitats will depend to a great extent on the matrix permeability (Ricketts 2001). Landscapes are commonly classified into continuous or fragmented (**Fahrig 2003)** however, the landscape is not a binary mosaic formed by natural habitat and matrix – or habitat and non-habitat – and the species certainly do not perceive it that way (presence/ absence of resources), as we will discuss later on in this chapter (**Fahrig, 2003)**

Despite being a rather controversial issue, several authors have shown the importance of protecting small native patches resulted from habitat fragmentation, as in the landscape context they are able to keep a significant portion of local biodiversity (Saunders *et al.* 1991; Lindenmayer & Nix 1993; Bodin et al. 2006). Andrèn (1994) states that landscape biodiversity may increase considerably when several small fragments are close to each other and permit animal and plant fluxes as in a continuous habitat. The ability of a species to move throughout the landscape is related to habitat connectivity (); it refers to the functional linkage among patches either due to patch proximity or to matrix permeability (With 1997; **Uezu** *et al.* 2005). Therefore, the degree of habitat connectivity, which depends on matrix quality, is essential to maintain native species in a fragmented landscape (Forman & Gordon 1986; With 1997; Tischendorf & Fahrig 2000; Ewers & Didham 2005).

Besides affecting species movements throughout the landscape, matrix quality also controls the permanence time of individuals in it, according to the resources it offers (Aberg *et al.* 1995), thus matrices of good quality may characterize a type of habitat effectively used by the species both in search for resources or traveling among preferential habitats (Smallwood & Fitzhugh 1995; Wagner & Fortin 2005). The quality of a matrix, however, is differently perceived by different species; some species may benefit from agricultural lands while others may be excluded (Gehring *et al.* 2003; Laurance 1994). As examples of the former case, some studies show the regular use of coffee plantations in Mexico (Moguel *et al.* 1999), banana and cocoa plantations in Costa Rica (Harvey *et al.* 2006), cocoa plantations in Brazil (Faria *et al.* 2006) and subsistence agriculture in Nepal (Acharya 2006) by the native fauna. On the other hand, species that required large territories and have small populations are particularly vulnerable to habitat fragmentation and can be locally extinct (Crooks 2002).

The increasing fragmentation and loss of natural areas, associated to changes in ecological processes and species extinctions demand urgent integration of human needs and the preservation of essential ecosystem processes. Approaches focused on the interactions between nature and man should be the basis for a transition to a more sustainable agriculture (Bignal 1998). The challenge of achieving development in a sustainable way was first globally discussed in the Brundlandt Commission (World Commission on Environment and Development – WCED), in 1983, and it has become a central question ever since. However, practical sustainable actions are being implemented very slowly although numerous studies have shown that the preservation of many species could be ensured if agricultural systems incorporated ecological concepts. On the other hand, conservationists have tried to find ways to integrate human land uses and native fauna needs (Vandermeer *et al.* 1997; Bignal 1998).

In this chapter, we intend to show the use of both natural and agricultural (silvicultural) habitats by the native carnivore fauna, and to demonstrate the possibility to maintain these populations in a fragmented landscape, provided that some large native patches are left and the matrix is permeable to the native fauna.

## STUDY REGION

São Paulo, in southeastern Brazil, is the most developed and urbanized state in the country. Despite representing less than 3% of the Brazilian territory, São Paulo State accounts for more than one third of the national gross domestic product and more than 21% of the country’s population (IBGE 2007a; IBGE 2007b). The state’s rapid development started about two centuries ago, when a strong agricultural expansion was implemented, based especially on coffee cultivation, which by 1950-60 was replaced by industrial development and agribusiness. Today, the state’s economy is based on industrial products and export commodities such as biofuel from sugarcane, paper and beef cattle (Igari et al. 2009).

As an expected consequence, São Paulo has lost more than 90% of its natural habitats, originally composed of the Atlantic dense rainforest on the east (alongside the Atlantic Ocean), a complex of savanna formations (regionally named *Cerrado*) in the central part of the state, and seasonal forests to the west. This strong process of habitat fragmentation in the state resulted in a few remaining patches of native vegetation, usually small and isolated (Metzger & Rodriges 2008).

In the northeast of São Paulo State, where our study was carried out (Santa Rita do Passa-Quatro and Luiz Antônio municipalities: 21°31’15’’S - 47°34’42’’W; 21°44’24’’S - 47°52’01’’W), a similar pattern of land occupation described for São Paulo State was observed: from 1962 to 1992, the region lost 60% of its original vegetation cover due to agriculture expansion (Kronka et al. 1993), and since 1992 agriculture and forestry are still expanding in the region but at a much slower pace. However, what distinguishes this region from the rest of São Paulo state is some large remnants of *cerrado* and seasonal forest. Our study region comprises the largest cerrado patches of the state, which are protected as nature preserves: Cerrado Pé-de-Gigante (1,212.9 ha), located in the Vassununga State Park (Korman 2003) and the Jataí Ecological Station (9,010.7 ha) (Decree 47.096/SP, from 18/September/2002). In addition, there are also in the study region four patches of seasonal forest in the study region, also part of the Vassununga State Park (sizes ranging from 12.1 ha to 327.8 ha) (Korman 2003) (Figure 1).

Therefore, the present land cover in the study region comprises a mosaic of natural formations and extensive monocultures, especially sugar cane and *Eucalyptus* species (Figure 2).

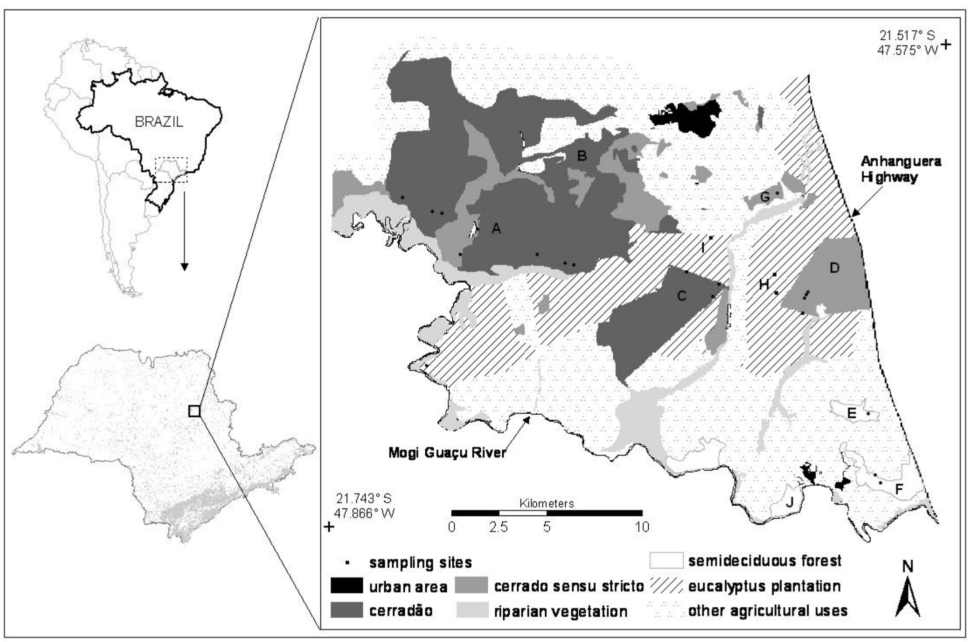


Figure 1. The study region (São Paulo State, Brazil): location and land use/ land cover classes. Dots represent the sampling sites; A, B and C = Jataí Ecological Station (EEJ) patches; D, E, F and J = Vassununga State Park (PEV) patches; G = private area with cerrado vegetation; H and I = eucalyptus plantation.


Figure 2. The mosaic of natural formations and extensive monocultures, especially of sugar cane and *Eucalyptus* species. (Photographed by Dr. Luciano Verdade)

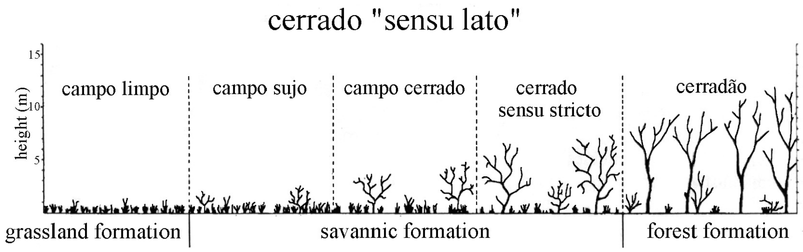


Figure 3. The *Cerrado* physiognomic gradient (modified from Coutinho 1978).

The native vegetation includes patches of seasonal forest and of different savanna physiognomies in an increasing density of trees, from *campo-sujo* (grassy field with scattered trees) to *cerradão* (sclerophyllous woodland), being *cerrado-sensu-stricto* an intermediate form (typical *cerrado* with grasses, shrubs and many trees) (Coutinho 1978; Oliveira & Marquis 2002; Shida 2005) (Figure 3).

The climate in the region is Cwa (according to Köppen 1948) or type II (following Walter 1986), which is the typical tropical savanna climate with wet summers (October to March) and dry winters (May to August); the annual rainfall is approximately 1,300 mm. The relief is gently rolling, formed by extensive and flat-topped hills.

## METHODS

To analyze the use of different habitats in the study region by the native fauna we sampled eight patches of native vegetation (three of *cerradão*, three of *cerrado-sensu-stricto*, and two of seasonal forest), as well as two homogeneous plantations of *Eucalyptus* species (Figure 1). We concentrated our analyses on mammal carnivores, as they are top predators and respond for several ecological processes in the community, therefore they may be used as indicator species of community resources and equilibrium (Crooks 2002; Miller et al. 2001).

Data were collected in three-day monthly field trips, throughout 18 months (August/2004 to January 2006). In the field, two systematic methods were used to obtain the data: camera trapping and track plot recording.

Track plot recording (Lyra-Jorge 1999; Pardini et al. 2003) is based on the identification of the animal species through footprints in a plot, and allows estimating animal occurrence and richness. We randomly selected 21 sampling points from a larger group of points that met the following requisites: located along pre-existing trails or dirt roads, and had previously recorded footprints in the soil, indicating that the animals effectively used that area. Twenty-one track plots of 10 m X 2 m were installed in the *Eucalyptus* plantation and native vegetation patches; the number of plots in the patches was proportional to their sizes, resulting in nine plots in *cerradão*, six in *cerrado-sensu-stricto*, two in seasonal forest and four in the *Eucalyptus* plantations (Figure1). The sandy ground was used to create each track plot. The track plots were visited every day, during the field trips, in order to identify the footprints in the soil and to clear the ground for new records. Ambiguous footprints were ignored and footprints of the same species, in the same plot, and in the same day were considered as if they were from a single individual.

The camera trapping method (Wemmer *et al.* 1996; Tomas & Miranda 2003) is based on the identification of the animal species through photographs taken by an automatic camera triggered by the animal body heat and/or movement. It also permits to estimate animal occurrence and richness. To distribute the camera traps in the area, we used the same criteria as those used to place track plots. In addition, cameras were protected from direct sunshine (as they would set off if exposed to intense heat). The sampling points were visited every field trip to change the films and batteries of the cameras, which remained activated in the field during the entire sampling period.

We assumed that all sampling points containing camera traps and track plots were homogeneous in detecting carnivores, and that all carnivore species were equally detectable by both methods.

We used data from both methods to calculate a species accumulation curve in order to express carnivore richness. The curves were randomized 5,000 times through a rarefaction process (Santos 2003). Species richness was also estimated through the Bootstrap technique (Smith & Van Belle 1984; Santos 2003) with 5,000 randomizations.

The relative frequency (FR) of the carnivores recorded by camera traps and track plots was calculated according to the model (Crooks 2002):

i/N , where: i = number of occurrences of species I;
N= total occurrences in the physiognomy.

We compared the distribution of species records in the *cerradão*, *cerrado-sensu-stricto* and *Eucalyptus* plantation through the Kruskal-Wallis test (Zar 1999); we removed the seasonal forest from this analysis due to the very small sample size (N=2). We also calculated and compared species diversity in each vegetation physiognomy (except for seasonal forest, due to small sample size) with the Shannon-Wiener diversity index (Magurran 1988) and Kruskal-Wallis test (Zar 1999).

The similarity in the carnivore assemblage in each vegetation form was tested through a Multi-Response Permutation Procedure (MRPP) analysis (McCune 2002) using data randomized 1,000 times and the Bray-Curtis index (Beals 1984). We used the numbers of species records in each sampling point.

To verify a possible influence of the landscape structure on the intensity of habitat use by the carnivores we created a carnivore habitat use map based on a model generated by stepwise regression analyses using the data on species richness and species occurrence obtained only through camera trapping, as well as information on habitat type (land use/ land cover class) and landscape structure. We assumed a direct correlation between the number of animal occurrences and the intensity of habitat use. We generated a land use/land cover map from a Landsat5-TM satellite image (February/2005, spatial resolution of 30 m) to obtain the landscape indices, and we then located the sampling points on that map and calculated the following landscape metrics: percentage of remaining habitat in the landscape (PLAND), edge density (ED) and habitat patch shape (SHP), in a multi-scale approach (Wagner & Fortin 2005; Fortin & Dale 2005; McAlpine *et al*. 2006), as for each point (and for each pixel) we used search radii of 250, 500, 1,000 and 2,000 m (following Umetsu *et al*. 2008). Landscape metrics were chosen according to our perceptions on the animals ecological needs based on field experience. They were calculated using the moving windows option of FRAGSTATS software (McGarigal & Marks 1995).

The response variable (habitat use intensity) was analyzed through stepwise regression. The animal occurrences were set as a dependent variable, while the metrics (PLAND, ED and SHP) of the four land use/land cover classes (*cerrado sensu stricto*, *cerradão*, seasonal forest and *Eucalyptus* plantation) in the four scales (radii of 250, 500, 1,000, and 2,000 m) were the independent variables. The stepwise regression analysis followed the general model below:

$$HUI = \beta1*M(d,c1)+ \beta2*M(d,c2)+ \beta3*M(d,c3)+ \beta4*M(d,c4)+ \varepsilon$$

where: *HUI* = habitat use intensity; *β1...β4* = regression parameters; *M(d,ci)* = landscape metrics (PLAND, ED and SHP) calculated for the distance *d* (specific radius scale) and the land use/land cover class *ci* (*cerrado-sensu-stricto*, *cerradão*, seasonal forest and *Eucalyptus* plantation); *ε* = error

The inclusion of each independent variable in the model was determined based on its statistically significant additional contribution.

The best model was the one with the highest coefficient of determination ($R^2$), and it was used to generate the map of habitat use intensity. To produce the map, we used the algebra map option of SPRING GIS (Cordeiro *et al*. 2008), where degrees of habitat use intensity were associated to color intensity.

## Results and Discussion: Habitat Use by Carnivores in the Fragmented Landscape

During the 18 months of sampling with both track plots (1,864 hours of exposure) and camera traps (12,960 hours of exposure), we were able to record ten carnivore species in the study region, which belonged to four different families (Table 1). Nine of these ten species were recorded by camera traps and seven species were recorded in the track plots. Two species of small felines (*Leopardus tigrinus* and *Puma yagouaroundi*) could not be distinguished by track plot recording, as their footprints were very much alike, but they could be recognized in the camera trap photographs. For this reason, these two species were grouped as "small felines" in some analyses. *Procyon cancrivorus* was not recorded by camera traps and was only found in the seasonal forest, while *Nasua nasua* was not recorded in the track plots (Table 1).

We believe we obtained a good representation of the local carnivore species richness, as the species accumulation curve calculated based on data from both methods approached the asymptote after nine months of sampling, and the obtained species richness (N= 9.0) was similar to the value estimated by Bootstrap (N= 9.16 ±0.16).

The species composition of the carnivore assemblage found in our study is in accordance with the species geographic distributions cited in the literature (Emmons 1997; Einsenberg & Redford 1999) and also agrees with other surveys carried out in the same region (Gargaglioni *et al.* 1998*;* Lyra-Jorge 1999; Talamoni *et al.* 2000). However, some carnivore species expected to be found in the region (according to Emmons 1997; Gargaglioni *et al.* 1998*;* Einsenberg & Redford 1999; Lyra-Jorge 1999; Talamoni *et al.* 2000) could not be detected. This could be either because some of those species are presently rare or even extinct in the region, or the methods used to sample local richness were not directed to some types of habitats and species niches. For example, *Lycalopex vetulus* and *Leopardus wiedii* – not detected – are naturally rare species, and their population densities are usually very low (Azevedo 1996; Jácomo *et al.* 2004); *Panthera onca* has not been seen in the region for more than 50 years and is probably locally extinct, as it was a favorite game animal; *Lontra longicaudis* and *Galictis cuja* are species associated to aquatic habitats, therefore their habitats or territories were not possible to be sampled with the methods we used (Emmons 1997; Pardini 1998).

*Puma concolor* and *Chrysocyon brachyurus* had the highest relative frequencies regardless of the sampling method (table 1), and there are several possible explanations for that. First, they are the largest carnivores in the region, have large home ranges and move constantly in search for food (Dietz 1984; Dickson & Beier 2002), and because of their vagility the same individuals might have been repeatedly recorded by the camera traps. Also, it has been shown that camera traps perform better in detecting large bodied animals (Carbone *et al.* 2002; Silveira *et al.* 2003) and consequently, these large animals may have been oversampled by the methodology here adopted (Lyra-Jorge *et al.* 2008).

An initial comparison of the habitat use by the species sampled through camera traps and track plots in *cerradão*, *cerrado-sensu-stricto*, and *Eucalyptus* plantation shows that although most species seem to prefer some habitats – *L. pardalis*, *N. nasua* and *C. semistriatus* were more frequent in *cerrado* physiognomies, whereas *C. thous* and *E. barbara* were more frequently found in the *Eucalyptus* plantation (table 1) – the distribution of all species records in these three vegetation forms was not statistically different (Kruskal-Wallis; p=0,943). This result was confirmed by the MRPP, which showed similarity in species composition among the different habitats (p=0.65; A=-0.013; expected $\Delta$= 0.51; observed $\Delta$= 0.52). Species diversity assessed by Shannon-Wiener index also showed no significant differences among those three habitat types (Kruskal-Wallis; p=0.31).

**Table 1. Carnivore species recorded by camera traps and in track plots. (NR= number of species records in all vegetation forms; CD= *cerradão*, SS= *cerrado-sensu-stricto*, SF= seasonal forest, EP= *Eucalyptus* plantation; FA= relative frequency obtained through camera trap data; FC= relative frequency obtained based on track plot data; FT= relative frequency obtained with both sampling methods.)**

| Species | Family | NR | Percentage of NR in the vegetation form | | | | Relative frequency (%) | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | CD | SS | SF | EP | FT | FA | FC |
| *Puma concolor* (Linnaeus, 1771) | Felidae | 74 | 49 | 22 | 9 | 20 | 29.4 | 30.1 | 29.0 |
| *Leopardus pardalis* (Linnaeus, 1758) | Felidae | 39 | 56 | 33 | 3 | 8 | 15.5 | 23.1 | 11.1 |
| Small cat | Felidae | 11 | 0 | 55 | 27 | 18 | 4.4 | 4.0 | 8.0 |
| *Chrysocyon brachyurus* (Illiger, 1811) | Canidae | 78 | 47 | 18 | 8 | 27 | 31.0 | 24.1 | 32.3 |
| *Cerdocyon thous* (Linnaeus, 1716) | Canidae | 27 | 37 | 15 | 7 | 41 | 10.7 | 3.2 | 12.0 |
| *Nasua nasua* (Linnaeus, 1766) | Procyonidae | 02 | 50 | 50 | 0 | 0 | 0.8 | 5.9 | 0.0 |
| *Procyon cancrivorus* (Cuvier, 1798) | Procyonidae | 03 | 0 | 0 | 100 | 0 | 1.2 | 0.0 | 4.1 |
| *Conepatus semistriatus* (Boddaert, 1784) | Mephitidae | 11 | 64 | 27 | 0 | 9 | 4.4 | 6.1 | 1.8 |
| *Eira barbara* (Linnaeus, 1758) | Mustelidae | 07 | 29 | 0 | 14 | 57 | 2.8 | 3.0 | 1.8 |
| *Total* | | 252 | | | | | 100.2 | 99.5 | 100.1 |

Similarly to the reported by other authors (Chinchila 1997; Oliveira 1998; Nuñez *et al.* 2000; Jácomo *et al.* 2004), these results indicated that even though most species maintained peculiar habitat preferences, the carnivore community as a whole was similar in the study region. This result supports the idea that carnivores in fragmented landscapes are more generalists than populations living in continuous and preserved areas and explore the entire region, not being restricted to the native vegetation patches (Azevedo 1996; Franklin *et al.* 1999; Donadio *et al.* 2001).

However, when we added information on the size and spatial arrangement of the different vegetation patches in the landscape the results were rather different. Although this analysis was performed using only camera trap data (Table 2), the species ratio recorded in the physiognomies were comparable in both data collecting methods. The model which included landscape structure data, vields high predictive power, with a significant coefficient of determination ($R^2 = 0.88$; $p < 0.01$). The best model selected took into account the occurrences of all the nine species recorded by camera traps (table 2), the four sampled habitats (*cerrado-sensu-stricto*: SS, *cerradão*: CD, seasonal forest: SF, *Eucalyptus* plantation: EP), the three landscape metrics (percentage of the habitat in the landscape: PLAND, edge density: ED, and patch shape: SHP), and only the radius of 250 m, as the independent variables generated on other scales (i.e., radii of 500, 1,000 and 2,000 m) did not make a statistically significant contribution to the model:

$$HUI = 0.04219*PLAND_{CD} + 0.01174*PLAND_{SS.} + 5.04857*SHP_{EP} - 0.26911*ED_{EP}$$

The independent variable that most contributed to the model was the percentage of *cerradão* in the landscape, $PLAND_{CD}$ (partial $R^2 = 0.50$; $p < 0.01$).

The map of habitat use generated from the model above shows the *Eucalyptus* plantation as the most intensely used habitat, followed by *cerradão*. Seasonal forest, on the other hand, was the least used habitat (Figure 4). Therefore, although we had noticed habitat preferences by some species when using exclusively information about land use/ land cover classes, these preferences only became detectable when considering landscape parameters were included, which refined the analysis. This indicates the importance of considering spatial parameters to build more accurate predictive models especially aimed at fauna conservation, as also noticed by other authors (Andrén 1994; Fahrig 1998 McAlpine *et al*. 2006).

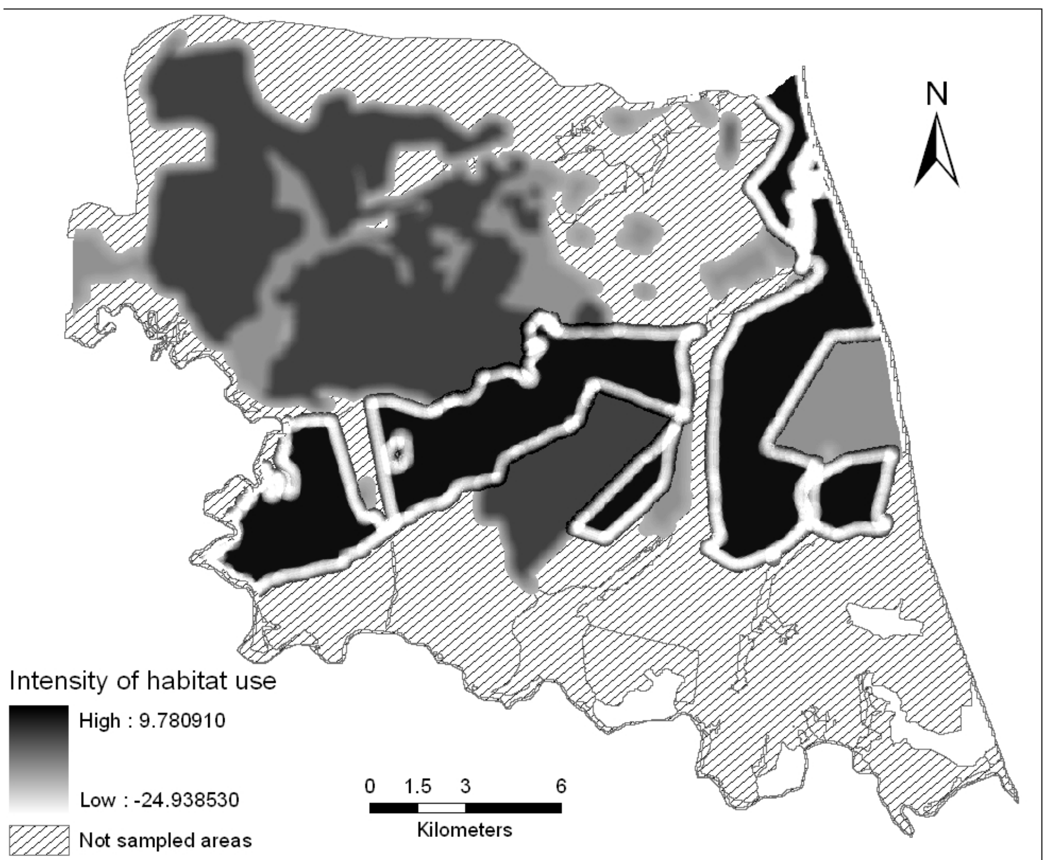


Figure 4. Intensity of habitat use by mammal carnivore species in the study region (São Paulo State, Brazil). (Darker to lighter = higher to lower intensity of use; see legend of vegetation physiognomies in Figure 1).

**Table 2. Carnivorous species recorded by camera traps in the study region (CD= *cerradão*, SS= *cerrado-sensu-stricto*, SF= seasonal forest, EP= *Eucalyptus* plantation).**

| Species | Family | Land use/ land cover class | | | | Total records |
|---|---|---|---|---|---|---|
| | | *CD* | *SS* | *SF* | *EP* | |
| *Puma concolor* | *Felidae* | 7 | 2 | 1 | 8 | 18 |
| *Leopardus pardalis* | *Felidae* | 9 | 2 | 0 | 2 | 13 |
| *Puma yagouaroundi* | *Felidae* | 0 | 0 | 0 | 1 | 1 |
| *Leopardus tigrinus* | *Felidae* | 0 | 1 | 0 | 0 | 1 |
| *Chrysocyon brachyurus* | *Canidae* | 7 | 1 | 0 | 6 | 14 |
| *Cerdocyon thous* | *Canidae* | 2 | 0 | 0 | 0 | 2 |
| *Eira barbara* | *Mustelidae* | 1 | 1 | 0 | 2 | 4 |
| *Conepatus semistriatus* | *Mephitidae* | 3 | 1 | 0 | 0 | 4 |
| *Nasua nasua* | *Procyonidae* | 1 | 1 | 0 | 0 | 2 |
| *Total* | | 30 | 9 | 1 | 19 | 59 |

Figure 5. Carnivores mammals in the study region. 1: Leopardus pardalis; 2: Eira Barbara; 3: Chrysocyon brachyurus; 4: Conepatus semistriatus; 5: Puma concolor.

The *Eucalyptus* stands in the study region are surrounded by large protected patches of *cerradão* (Jataí Ecological Station-EEJ), and both the silviculture and the *cerradão* hold similarities in terms of vegetation structure (arboreal habitats with not very open canopy, sparse herbaceous layer, etc). Moreover, *Eucalyptus* cultures are based on 7-year cycles and are not as much intensely managed as the other crops in the region (ex. sugar-cane). We can then infer that carnivore species use *Eucalyptus* plantations at least to move around, although we do not know what other resources this vegetation may offer to the carnivores. Some generalist species are able to adapt to man-modified environments when their original habitats are severely reduced, and to find vital resources in such new habitats (Sánchez-Hernandéz et al. 2001; Reznick *et al.* 2004; Tabeni *et al.* 2005; Morán-López *et al.* 2006; McDougall *et al.* 2006). In this study region, *Puma concolor*, *Chrysocyon brachyurus*, *Cerdocyon thous* and *Eira barbara* seem to be relatively adapted to silvicultural *Eucalyptus* plantations, as they frequently use them, confirming the reported by other authors (Bisbal 1986; Jácomo *et al.* 2004).

*Cerradão*, the native physiognomy most intensely used by carnivores, comprises the largest native vegetation patches in the study region, all protected as natural preserves. These patches in the study region, all protected as natural preserves. These patches are mostly contiguous to the Mogi-Guaçu River riparian forests and swamps. These features may represent good quality habitats for the local fauna. On the other hand, seasonal forest patches are small and surrounded by sugar-cane plantations, which are highly managed throughout the year and receive large quantities of pesticides. Similarly, *cerrado sensu stricto* patches are small (except from Vassununga State Park, PEV, Figure 1 D) and surrounded by sugar-cane fields.

Therefore, our data show that top predator carnivores of medium to large size can be maintained in a highly fragmented agricultural landscape provided some large patches of native vegetation in good condition remain (performing as source patches, according to Donavan *et al.* 1995), and they are surrounded by a permeable matrix. Other authors came to similar conclusions in relation to other animal groups elsewhere (McAlpine *et al.* 2006; Baldissera *et al*. 2008; Marsden & Symes 2008). A highly permeable matrix is essential to connect habitats, permitting animal movements throughout the landscape and the maintenance of processes that are essential to the populations' persistence, such as dispersion and gene flow, in addition to animal foraging and housing (Elmhagen & Angerbjörn 2001; Hensen *et al.* 2005). In this study region, *Eucalyptus* plantations act as permeable matrix connecting patches of native vegetation, and this may be the reason for the permanence of a still rich carnivore assemblage in such an agricultural landscape (Lyra-Jorge & Pivello 2005).

## CONCLUSION

A landscape is comprised of different types of habitats, used by fauna with different objectives – from foraging to reproduction – according to resource availability and quality. Habitats more intensely used are generally those with enough quality to maintain species residence and/or movement, and they usually either offer a great deal of vital resources or connect areas that have such resources, in that case acting as biodiversity corridors. In this sense, the manner and intensity of use of remnant habitats by native fauna is a relevant issue for biodiversity conservation.

In a scenario where it is no longer possible to exclude the intensive use of land by man and to keep large preserved natural areas, it is fundamental to contemplate means of turning agricultural matrices into good quality habitats for fauna. We showed in this chapter that it is possible to balance human land uses and fauna requirements provided that some ecological principles are taken into account, especially related to landscape structure. In the case of large and medium carnivores – as they are vagile animals that explore large territories – the maintenance of a net system of protected areas of different sizes that keep connectivity through a permeable matrix is essential.

## REFERENCES


Acharya, K. P. Linking tree on farms with biodiversity conservation in subsistence farming in Nepal. *Biodiversity and Conservation, 15*, 631-646.

Andrén, H. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: A Review. *Oikos, 71(3)*, 355-366.

Arberg, J., Jansson, G., Swenson, J. E. & Angelstam, P. (1995). The effect of matrix on the occurrence of hazel grouse in isolated habitat fragments. *Oecologia, 103*, 265-269.

Azevedo, F. C. C. (1996). Notes on behavior of the margay *Felis wiedii* in the Brazilian Atlantic forest. *Mammalia, 60*, 325-328.

Baldissera, R., Ganade, G., Brescovit, A. D. & Hartz, S. M. (2008). Landscape mosaic of Araucaria forest and forest monocultures influencing understorey spider assemblages in southern Brazil. *Austral Ecology* 33: 45-54. BEALS, E.W. 1984. Bray-Curtis ordination: A strategy for analysis of multivariate ecological data. *Advances in Ecological Research, 14*, 1-56.

Bignal, E. M. (1998). Using an ecological understanding of farmland to reconcile nature conservation requirements, EU agriculture policy and world trade agreements. *Journal of Applied Ecology, 35*, 949-954**.**

Bisbal, F. J. (1986). Food habits of some neotropical carnivores in Venezuela. *Mammalia, 50(3)*, 329-339.

Bodin, O; Tengö, M., Norman, A., Lundberg, J. & Elmquist, T. (2006). The value of small size: Loss of forest patches and ecological thresholds in southern Madagascar. *Ecological Applications, 16(2)*, 440-451.

Carbone, C., Conforti, C., Coulson, T., Franklin, N., Ginsberg, J. N., Grivths, M., Holden, J., Kinnaird, M., Laidlaw, R., Lynam, A., MacDonald, D. W., Martyr, D., McDougal, C., Nath, L., O'Brien, T. O., Seidnsticker, J., Smith, D. J., Tilson, R. & Shahruddin, W. N. (2002). The use of photograph rates to estimate densities of tigers and other cryptic mammals: response to Jannelle *et al. Animal Conservation, 5*, 121-132.

Chinchila, F. A. (1997). La dieta del jaguar (*Panthera onca*), el puma (*Felis concolor*) y el manigordo (*Felis pardalis*) em el Parque Nacional Corcovado, Costa Rica. *Ver Biologica Tropical, 459*, 1223-1229.

Cordeiro, J. P. C., Câmara, G., Freitas, U. M. & Almeida, F. (2008). Yet Another Map. *GeoInformatica*, DOI 10.1007/s10707-008-0045-4.

Coutinho, L. M. (1978). O conceito de cerrado. *Revista Brasileira de Botânica*, 1, 115-117.

Crooks, K. (2002). Relative sensitivities of mammalian carnivores to habitat fragmentation. *Conservation Biology, 16*, 488-502.

Dickson, B. G. & Beier, P. Home range and habitat selection by adult cougars in southern California. *Journal of Wildlife Management, 66(4)*, 1235-1245.

Dietz, J. (1984). Ecology and social organization of the maned wolf (*Chrysocyon brachyurus*). *Smithsonian Contribution Zoology, 392*, 1-51.

Donadio, E., Di Martino, S., Aubone, M. & Novaro, A. J. (2001). Activity patterns, home range, and habitat selection of the common hog-nosed skunk, *Conepatus chinga* in northwestern Patagonia. *Mammalia, 65*, 49-54.

Donavan, T., Lamberson, R., Kimber, A., Thompson, R. & Faaborg, J. (1995). Modeling the effects of habitat fragmentation on source and sink demography neotropical migrant birds. *Conservation Biology, 9(6)*, 1397-1407.

Eisenberg, J. F. & Redford, K. H. (1999). *Mammals of the Neotropics***.** University of Chicago Press.USA. 609.

Elmhagen, B. & Angerbjörn, A. (2001). The applicality of metapopulation theory to large mammals. *Oikos, 94*, 89-100.

Emmons, L. (1997). *Neotropical rainforest mammals: a field guide.* University of Chicago Press. USA. 307.

Ewers, R. & Didham, R. (2006). Confounding factors in the detection of species responses to habitat fragmentation. *Biological Reviews, 81*, 117-142.

FAHRIG, L. 1998. When does fragmentation of breeding habitat affect population survival? *Ecological Modelling, 105*, 273-92.

Faria, D., Laps, R. R., Baumgarten, J. & Cetra, M. (2006). Bat and bird assemblages from forests and shade cacao plantations in two contrasting landscapes in the Atlantic forest of southern Bahia, Brazil. *Biodiversity and Conservation, 15*, 587-612.

Farigh, L. (2003). Efffects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution, and Systematics, 34*, 487-515.

Forman & Gordon (1986). *Landscape Ecology.* John Willey and Sons.

Fortin, M. J. & Dale, M. R. T. (2005). *Spatial Analysis: A Guide for Ecologists.* Cambridge University Press, Cambridge.

Franklin, W., Johnson, W., Sarno, R. & Iriarte, J. A. (1999). Ecology of the Patagonia puma in southern Chile. *Biological Conservation, 19*, 33-40.

Gargaglioni, L. H., Batalhão, M. E., Lapenta, M. J., Carvalho, M. F., Rossi, R. V. & Veruli, V. P. (1998). Mamíferos da Estação Ecológica de Jataí, Luiz Antônio, SP. *Papéis Avulsos de Zoologia, 40*, 267-287.

Gascon, C., Lovejoy, T. E., Bieregaard, R. O., Malcom, J. R., Stouffer, P. C., Vasconcelos, H., Laurance, W., Zimmerman, B., Tocher, M. & Borges, S. (1999). Matrix habitat and species richness in tropical Forest remnants. *Biological Conservation, 91*, 223-229.

Gehring, T. M. & Swihart, R. K. (2003). Body size, niche breath, and ecologically scaled responses to habitat fragmentation: mammalian predators in an agricultural landscape. *Biological Conservation, 109*, 283-295.

Harvey, C., Gonzalez, J. & Somarriba, E. Dung beetle and terrestrial mammal diversity in forests, indigenous agroforestry systems and plantain monocultures in Talamanca, Costa Rica. *Biodiversity and Conservation, 15*, 555-585.

Hensen, A., Knight, R. L., Marzluff, J. M., Powell, S., Brown, K., Gude, P. &, Jones, K. (2005). Effects of exurban development on biodiversity: patterns mechanisms and research needs. *Ecological Application, 15*, 1893-1905.

I.B.G.E. (2007a). Contas nacionais – Produto interno bruto dos municípios 2002-2005. Instituto Brasileiro de Geografia e Estatística - IBGE, Rio Janeiro, Brasil, 230.

I.B.G.E. (2007b). Contagem da população 2007. Instituto Brasileiro de Geografia e Estatística - IBGE, Rio Janeiro, Brasil, 311.

Igari, A. T., Tambosi, L. R. & Pivello, V. R. (2009). In press. Agribusiness opportunity costs and environmental legal protection: Investigating this trade-off on a hotspot preservation in the State of São Paulo – Brazil. Environental Management.

Jácomo, A. T., Silveira, L. & Diniz-Filho, A. F. (2004). Niche separation between the maned wolf (*Chrysocyon brachyurus)*, the crab-eating fox (*Dusicyon thous*) and the hoary fox *(Dusicyon vetulus*) in Central Brazil. *Journal Zoology of London, 262*, 99-106.

Korman, V. (2003). *Proposta de interligação das glebas do Parque Estadual de Vassununga (Santa Rita do Passa Quatro, SP).* MSc Thesis. Escola Superior de Agricultura Luiz de Queiroz, Universidade de São Paulo, Piracicaba, SP., Brazil.

Kronka, F. J. N., Matsukuma, C. K., Nalon, M. A., Del Cali, I. H., Rossi, M., Matos, I. F. A., Shin-Ike, M. S. & Pontinhas, A. S. (1993). *Inventário Florestal do estado de São Paulo*. Instituto Florestal. São Paulo. 199.

Laurance, W. F. (1994). Rainforest fragmentation and the structure of small mammal communities in tropical Queensland. *Biological Conservation, 69*, 23-32.

Lindenmayer, D. B. & Nix, H. (1993). Ecological principles for the design of wildlife corridors. *Conservation Biology, 55*, 77-92.

Lyra-Jorge, M. C., Ciocheti, G. & Pivello, V. R. (2008). Carnivore mammals in a fragmented landscape in northeast of São Paulo State, Brazil. *Biodiversity and Conservation, 17*, 1573-1580.

Lyra-Jorge, M. C. & Pivello, V. R. (2005). Mamíferos. 135-148. In V. R. Pivello, & E. M. Varanda (Eds.), *O Cerrado Pé-de-Gigante (Parque Estadual de Vassununga, São Paulo) - Ecologia e Conservação.* São Paulo, Secretaria de Estado do Meio Ambiente.

Lyra-Jorge, M. C. (1999). *Avaliação do potencial faunístico da A.R.I.E. Cerrado Pé-de-Gigante (Parque Estadual do Vassununga, Santa Rita do Passa Quatro – SP) com base na análise de habitats*. MsC thesis. Instituto de Biociências, Universidade de São Paulo.São Paulo, SP, Brazil.

Magurran, A. E. (1998). *Ecological diversity and its measurement*. University Press, Cambridge. UK. 178.

Marsden, S. J. & Symes, C. T. (2008). Bird richness and composition along an agricultural gradient in New Guinea: The influence of land use, habitat heterogeneity and proximity to intact forest. *Austral Ecology, 33*, 784-793.

Mcalpine, C. A., Bowen, M. E., Callaghan, J. G., Lunney, D., Rhodes, J. R., Mitchell, D. L., Pullar, D. V. & Poszingham, H. P. (2006). Testing alternative models for the conservation of koalas in fragmented rural–urban landscapes. *Austral Ecology, 31*, 529-544.

McCune, B. & Grace, J. B. (2002). *Analyses of ecological communities*. Software design, Gleneden Deach. USA.

McDougal, P. T., Réale, D., Sol, D. & Reader, S. M. (2006). Wildlife conservation and animal temperament: causes and consequences of evolutionary change for captive reintroduced and wild populations. *Animal Conservation, 9*, 39-48.

McGarigal, K. & Marks, B. (1995). *FRAGSTATS: spatial pattern analysis. Program to quantifying landscape structure.* Department of Agriculture, Forest Service. USA. 122.

Miller, B., Dugelby, B., Foreman, D., Rio, C. M., Noss, R., Phillips, M., Reading, R., Soulé, M. E., Terborgh, J. & Willcox, L. (2001). The importance of large carnivores to healthy ecosystems. *Endangered Species UPDATE, 18(5)*, 202-210.

Moguel, P. & Toledo, V. M. (1999). Biodiversity conservation in traditional coffee systems of Mexico. *Conservation Biology, 13(1)*, 11-21.

Morán-López, R., Guzmán, J. M., Borrego, E. C. & Sánchez, A. V. (2006). Nest-site selection of endangered cinereous vulture populations affect by anthropogenic disturbance: present and future conservation implications. *Animal Conservation, 9*, 29-37.

Nagendra, H., Munroe, D. & Southworth, J. (2004). From pattern to process: landscape fragmentation and the analysis of land use/land cover change. *Agriculture, Ecosystems & Environment, 101(2-3)*, 111-115.

Nuñez, R., Miller, B. & Lindzey, F. (2000). Food habits of jaguars and pumas in Jalisco, Mexico. *Journal of Zoology of London, 252*, 373-379.

Oliveira, P.S. & Marquis, R. J. (2002). *The cerrados of Brazil. Ecology and natural history of a neotropical savanna.* Columbia University Press, New York, 398.

Oliveira, T. G. (1998). *Herpailurus yagouaroundi. Mammalian Species, 578*, 1-6.

Pardini, R. (1998). Feeding ecology of the neotropical river otter *Lontra longicaudis* in an Atlantic forest stream, south-eastern Brazil. *Journal of Zoology of London, 245*, 385-391.

Pardini, R., Ditt, E. H., Cullen, L. JR., Bassi, C. & Rudran, R. (2003). Levantamento rápido de mamíferos terrestres de médio e grande porte. 181-202. In L. Cullen Jr., R. Rudran, & C. Valladares-Pádua, (Eds.), *Métodos de estudo em biologia da conservação e manejo de vida silvestre.* Editora da UFPR; Fundação O Boticário de Proteção à Natureza. Curitiba.

Reznick, D., Rodd, H. & Nunney, L. (2004). Empirical evidence for rapid evolution. In: R. Ferrière, U. Diekman, & D. Couvert, (Eds.), *Evolutionary Conservation Biology*. Cambridge University Press. Cambridge, U.K. 428.

Ricketts, T. (2001). The Matrix Matters: Effective Isolation in Fragmented Landscapes. *American Naturalist, 158*, 87-99. de alteración em el sureste de México. *Acta Zoológica del Mexico, 84*, 35048.

Sánchez-Hernandez, C., Romero-Almaraz, M. L., Colín-Martinez, H. & García-Estrada, C. (2001). Mamíferos de cuatro áreas com diferente grado de alteración em el sureste de México. *Acta Zoológica del Mexico, 84*, 35048.

Santos, J. A. (2003). Estimativa de riqueza em espécies. 19-42. In L. Cullen Jr., R. Rudran, & C. Valladares-Pádua (Eds.), *Métodos de estudo em biologia da conservação e manejo da vida silvestre.* Editora da UFPR. Curitiba.

Saunders, D. A., Hobbs, R. J. & Margulis, C. R. (1991). Biological consequences of ecossystem fragmentation: a review. *Conservation Biology, 5*, 18-32.

Shida, C. N. (2005). Caracterização física do cerrado Pé-de-Gigante e uso das terras na região. Evolução do uso das terras na região. 25-47 In V. R. Pivello, & E. aranda, (Eds.), *O Cerrado Pé-de-Gigante. Parque Estadual de Vassununga. Ecologia e Conservação.* SEMA, São Paulo.

Silveira, L., Jácomo, A. T. & Diniz-Filho, J. A. (2003). Camera trap, line transect census and track surveys: a comparative evaluation. *Biological Conservation, 114*, 351-355.

Smallwood, K. S. & Fitzhugh, E. L. (1995). A track count for estimating Mountain lion *Felis concolor californica* population trend. *Biological Conservation, 71*, 251-259.

Smith, E. P. & Van Belle, G. (1984). Non parametric estimation of species richness. *Biometrics, 40*, 119-129.

Tabeni, S. & Ojeda, R. A. (2005). Ecology of the desert small mammals in disturbed and undisturbed habitats. *Journal of Mammalogy, 70(2)*, 416-420.

Talamoni, S. A., Motta-Júnior, J. C. & Dias, M. M. (2000). Fauna de mamíferos da Estação Ecológica de Jataí e Estação Experimental de Luiz Antônio. 317-319. In J. E. Santos, &

J. S. R. Pires, (Eds.), *Estudos Integrados em Ecossistemas.Estação Ecológica de Jataí.* RiMa Editora. São Carlos. 346.

Tischendorf, L. & Fahrig, L. (2000). How should we measure landscape connectivity? *Landscape Ecology*, *15*, 633-641.

Tomas, W. M. & Miranda, G. H. B. (2003). Uso de armadilhas fotográficas em levantamentos populacionais. 243-268. In L. Cullen Jr., R. Rudran, & C. Valladares-Pádua (Eds.), Métodos de estudo em biologia da conservação e manejo da vida silvestre. Editora UFPR.

Uezu, A., Metzeger, J. P. & Vielliard, J. M. E. (2005). Effects of structural and functional connectivity and patch size on the abundance of seven Atlantic forest bird species. *Biological Conservation, 123(4)*, 507-519.

Umetsu, F., Metzeger, J. P. & Pardini, R. (2008). Importance of estimating matrix quality for modeling species distribution in complex tropical landscapes: a test with Atlantic forest small mammals. *Ecography*, *31*, 359-370.

Vandermeer, J. & Perfecto, I. (1997). The agroecosystem: a need for the conservation biologist's lens. *Conservation Biol*ogy, *11(3)*, 591-592.

Wagner, H. & Fortin, M. J. (2005). Spatial analysis of landscapes: Concepts and statistics. *Ecology, 86(8)*, 1975-1987.

Wemmer, C., Kunz, T., Lundie-Jekins, G. & McShea, W. (1996). Mammalian Sign. 157-176. In D. E. Wilson, F. R. Cole, J. D. Nichols, R. Rudran, & M. S. Foster, (Eds.), *Measuring and monitoring biological diversity. Standard methods for mammals.* Smithsonian Intitution Press.

Wilcove, D. S., McLellan, C. H. & Dobson, A. P. (1986). Habitat fragmentation in the temperate zone. In: Soulé, M. E. *Conservation Biology*. M. A. Sunderland, K. With, R. Gardner, & M. Turner (Eds.), Landscape connectivity and population distributions in heterogeneous environments. *Oikos, 78,* 151-169.

Zar, J. H. (1999). *Biostatistical analysis*. Prencinton Hall, New Jersey. 409.