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Fragmentation of the Atlantic Rainforest in the Northern Coastal Region of Pernambuco, Brazil: Recent Changes and Implications for Conservation

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ABSTRACT

In the Atlantic Forest of Northeastern Brazil, habitat loss and fragmentation are the major threats to biodiversity. In the State of Pernambuco, where the landscape has been altered mainly for sugarcane cultivation, this study was carried out in a 280 km^2 area, using aerial photographs from 1975 and high-resolution satellite images from 2005, integrated into a GIS. Our main goals were to analyze spatial attributes and landscape changes, to quantify forest loss and isolation and to discuss implications for forest conservation. Analyses included all mature forest (MF) patches and non-forest areas in both years, and secondary forest (SF) patches in 2005. Landscape metrics and deforestation rate were calculated, and all patches were assigned to size and shape categories. Between 1975 and 2005, MF cover and patch number decreased from 45.61% (153) to 24% (110), with a mean annual forest loss rate of 2.14%. Mean patch size decreased from 83.47 to 61.10 ha. Patch isolation increased, as mean nearest neighbor changed from 397.10 to 695.97m, isolation index increased from 78.22 m to 440.79 m, and distribution pattern changed from clustered to random. Number of small fragments and proportion of irregular shaped patches varied from 108 to 70, and 74.5 to 72.7%, respectively, with a significant difference in shape classes. Altogether, these results picture a pattern of patch shrinking. In the present landscape, 96 SF patches were identified, 76% of which originated from mature forests. Although the fragmentation pattern in the studied area is very critical for forest conservation, this study points out for simple and effective conservation actions, such as protection of large fragments, the union of small to large fragments through reforestation, the creation of forest corridors, and the maintenance of SF patches, all executed in partnership with the private sugar-alcohol sector.

Keywords: aerial photographs, deforestation, GIS, Ikonos II, landscape ecology, metrics

INTRODUCTION

Ecosystem fragmentation, habitat loss and landscape modification are major threats to natural systems and biodiversity (Tabarelli and Gascon 2005; Fischer and Lindenmayer 2007). While fragmentation is the breaking apart of a habitat, changing its configuration, independent of area loss, habitat loss is the removal of habitats (Fahrig 2003), with a consequent reduction in cover. These two processes occur together, and are dependent on the purposes and historical aspects that drive landscape modification, and on the frequency and intensity of human impacts. As a result, a modified landscape has fragments of various sizes, shapes, ages and patterns of connectivity and its dynamics and modification rates, along with the sensitivity of species to these factors, will determine the survivorship of biological communities.

The various processes that modify the landscape have many biological consequences at patch or landscape level. In the remnants, edge effects play a crucial role, and are described as biotic and abiotic changes along remnant margins creating microclimatic conditions different from interior habitats (Young and Mitchell 1994; Murcia 1995). Area effects also occur, and refer to ecological changes due to isolation, proportional to fragment area (Nascimento and Laurance 2006). Both processes influence species richness and composition, population and community structure, ecological processes, such as herbivory, predation, competition and parasitism (Ewers and Didham 2006) and genetic variability (Kageyama and Gandara 1998). On a landscape level, studies suggest that a disproportionate loss of species occurs when total habitat cover decreases to a critical percentage of the landscape (Radford *et al.* 2005) or when landscape changes from an intact to a relictual configuration, with less than 10% of remaining cover (Fischer and Lindenmayer 2007).

The importance of spatial interactions between landscape units and ecological patterns and processes, and the relevance of such relationships for the biological conservation are the main focus of landscape ecology (Metzger 2001). Landscape pattern indices or metrics provide a useful tool to explore insite variability (Echeverría *et al*. 2006), as well as for the assessment of landscape dynamics (Ferraz *et al.* 2005). Over the last decades, landscape ecology has promoted a change on the studies of fragmentation and conservation of species and ecosystems, allowing a greater resolution of the environmental problems. Studies that approach landscape dynamics, providing estimates of deforestation and ecosystem transformation, are of great value for understanding the consequences of human activities on the natural environment (Armenteras *et al.* 2006).

Spatial studies of forest fragments provide information that can subsidize conservation actions and mitigation planning, for the maintenance of biological diversity in fragmented landscapes and for restoring connectivity, which reduces risks of local extinction and favors re-colonization

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(Metzger 2003a). Many studies were carried out recently to evaluate the fragmentation and habitat loss of Brazilian ecosystems in the Amazon, Cerrado and Atlantic Rainforest. Most studies of landscape structure, however, consider only a single, specific time (Jorge and Garcia 1997; Tanizaki and Moulton 2000; Saatchi *et al.* 2001; Ditt 2002; Martins *et al.* 2002; Becker *et al.* 2004; Durigan *et al.* 2007). Although landscapes are affected by ongoing processes (Ferraz *et al.* 2005), few studies analyzed temporal changes, and these are concentrated in the Amazon (Sierra 2000; Metzger 2002; Ferraz *et al.* 2005; Armenteras *et al.* 2006).

The Brazilian Atlantic Rainforest, which originally covered 1.3 million km^2, is a key example of the processes of fragmentation and habitat loss, mainly for timber, firewood, charcoal, agriculture, cattle ranching, and the construction of cities, which reduced this forest to only 7.6% of its original cover (Morellato and Haddad 2000). In the Northeastern region, the situation is more critical, and the percentage of forest loss can reach 98-99% in some parts, most converted into agricultural land (Viana *et al.* 1997). Historically, many causes for the deforestation of the Atlantic Rainforest can be listed. One of the main causes was the clearing of large areas for cultivation of sugarcane, leaving behind small forest patches of irregular shapes. Preservation of most fragments is usually related to relief configuration, which does not favor anthropic occupation and soil utilization for economic purposes (Ranta *et al.* 1998; Silva *et al.* 2007, 2008). Despite its degree of threat, the Atlantic rainforest still contains a species diversity higher than most of the Amazon forests, a high species diversity and level of endemism (Morellato and Haddad 2000).

In the State of Pernambuco, an important and pioneer study of fragmentation examined the sizes, shapes and distribution of forest remnants in the southeastern corner of the state (Ranta *et al.* 1998). Among their main assumptions, Ranta *et al.* (1998) estimated that the decrease in rain forest area had been only about 10% during the 1980 and 1990 decades, when the government-subsidized 'Próalcool' programme promoted the cultivation of sugarcane. Besides, their study was carried out in 1990 decade, but was based on maps from 1974 and did not describe landscape dynamics; and they hypothesized that, as the Brazilian government stopped subsidizing sugar cane production by 1993, many sugarcane fields would have been abandoned and forests would regenerate. However, there are spatial and temporal changes in this scenario. In the Northeastern portion of the state, sugarcane fields still cover 80% of the agricultural lands (CPRH 2003); this decade has seen a crescent expansion of ethanol production from sugarcane envisaged in Brazil to supply an expanding market as well as exports to other countries (Goldemberg *et al.* 2008); and local informants in the northeastern Pernambuco estimate a forest loss much higher than 10% during the 'Próalcool' decade. On the good side, conservation practices and forest value also changed. Although most forest fragments are privately owned, incentive mechanisms and economic opportunities are now helping the protection and restoration of the remaining Atlantic Forest (Tabarelli *et al.* 2005) and concerns on the sustainability of ethanol production have been raised (Goldemberg *et al.* 2008).

In this study, landscape structure, including spatial distribution and characteristics of forest patches, were examined for two periods, using aerial photographs and high-resolution satellite imagery, in a landscape ecology approach. Our main goals were to analyze spatial attributes and landscape changes, to quantify forest loss and isolation and to discuss implications and future scenarios for the conservation of Atlantic rainforest remnants in the sugar-cane matrix.

MATERIALS AND METHODS

Study area

The northern region in the coastal area of the State of Pernambuco, Brazil, covers 1377 km^2, and includes eight municipalities (CPRH 2003). This study was carried out in a segment of this area covering 280 km^2, of which 88% are owned by a single property, the Usina São José (São José Sugar Mill), since 2001 (**Fig. 1**). The

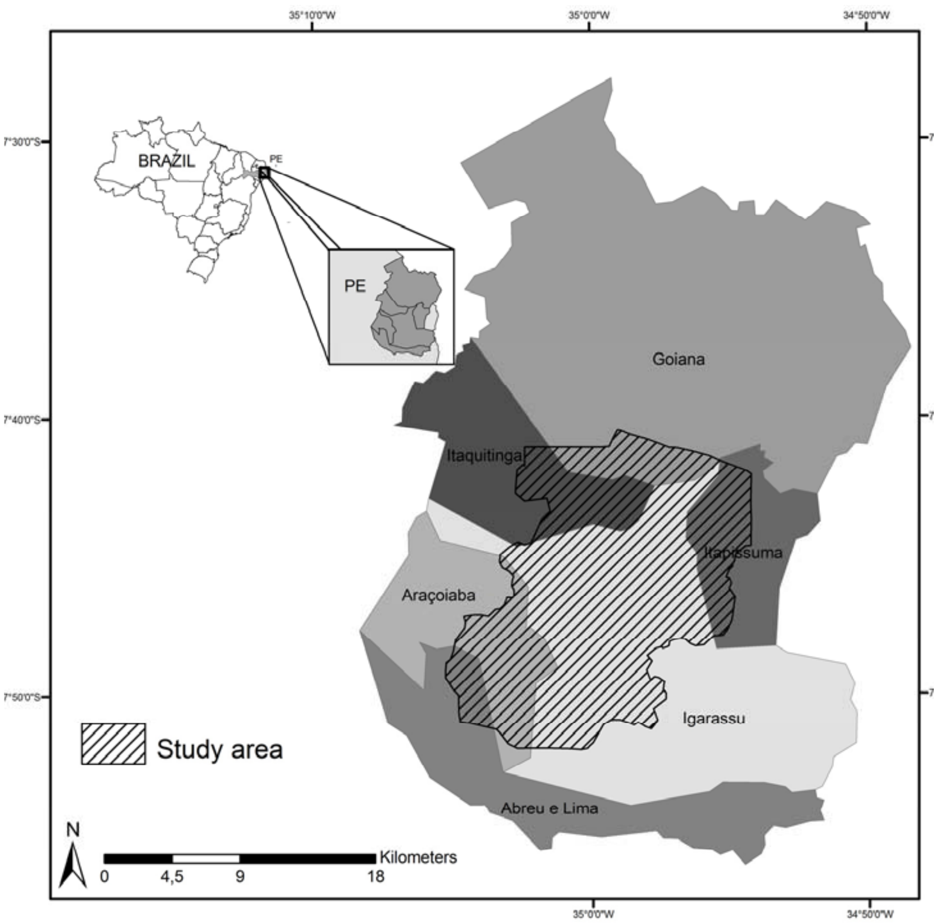

Fig. 1 Location of the study area, in the Northern region of the coast of Pernambuco, Brazil.

studied area is circumscribed between the geographic coordinates 07° 40′ 21.25″ and 07° 55′ 50.92″ S; 34° 54′ 14.25″ and 35° 05′ 21.08″ W, and includes a large portion of the municipality of Igarassu, as well as small areas of the municipalities of Goiana, Itaquitinga, Itapissuma, Abreu e Lima and Araçoiaba (**Fig. 1**).

The climate type is As' (Köppen 1936), hot and humid, characterized by a mean annual rainfall of 1687 mm and mean temperature of 24.9°C (meteorological data collected at the Usina São José Station, from 1998 to 2006). Hydrological superficial resources in the area include: part of the Botafogo-Arataca drainage basin, containing the only reservoir in the Northern Section – the Botafogo Dam; and the Santa Cruz Channel, which receives the water from many rivers and streams in the region (CPRH 2003).

The predominant geological feature in the region is the Barreiras Group, a continental terrigenous sedimentary deposit of Plio-Pleistocenic age, dominantly composed by sandy and clay sediments. The relief is composed by flat plateaus, excavated by narrow and deep valleys, with steep slopes, usually with declivity greater than 30%. Declivity in this region is the main restricting factor for land use, including urbanization and agricultural purposes. The potential vegetation is the dense ombrophilous Atlantic rainforest, but the landscape has been altered by human activities, mainly deforestation followed by sugarcane cultivation. In the present, farmed fields with sugarcane plantations are the predominant land use (CPRH 2003), occupying the plateaus and riparian terraces (flat surfaces), as well as the hilly areas with steep valleys. Forest clearance began with the arrival of Portuguese colonizers in the XVI century, at first for wood exploitation, secondly for cattle raising and later for sugarcane production (Câmara 2003). In recent decades, there was a boom in forest elimination for sugarcane cultivation for biofuel production, principally from middle 1970s to middle 1980s (Farias *et al.* 2007; Goldemberg *et al.* 2008).

In this region, there is only one protected area, the ecological state reserve "São José Forest" or "Mata de Piedade", established by the state law 9989 (1987), covering 305.78 ha of mature forests with a well preserved condition (Santos *et al.* 2001).

Data acquisition

Digitized maps from two different years (1975 and 2005) in the form of vector versions were used in this study. These data sets covered the entire polygon and were generated from the following data sources:

1) For 1975 data, orthophotographs obtained from the Foundation for the Development of the Metropolitan Region of Recife, Pernambuco (FIDEM), scale 1:10,000, were scanned and converted into panchromatic images (300 and 600 dpi), TIFF format. The cartographic projection system used was UTM (Zone 25, central meridian 33° 00′ 00″ W, horizontal datum SAD 69);

2) For 2005 data, digital multi-spectral satellite images from IKONOS II, with high spatial resolution (4 m) and radiometric resolution of 8 bits, UTM projection and Datum WGS 84; additionally, topographic (planimetric) maps in DWG format were obtained, converted to shapefile (scale 1:5,000), based on a reconnaissance field survey using a global positioning system with geodesic receptors and differential method.

Using software SPRING 3.6, TIFF files were converted to GEOTIFF, and geo-coded into the same geographic co-ordinates, using as control points a grid with geographic coordinates and ground points obtained from GPS.

Data preparation

Datasets originated a digital mosaic for the studied landscape in 1975 and 2005, integrated into a geographic information system (GIS) using the software ArcGIS 9.2 (ESRI 2006) and its extension Spatial Analyst. All patches greater than 1 ha of mature forest (MF) existent in both years were vectorized on screen. In addition, patches of secondary forest (SF) existent in 2005, independently of size, were also vectorized. SF patches from 1975 were not vectorized due to the difficulty in obtaining the precise separation between SF and plantations based on the aerial photographs. Field surveys for ground truthing were carried for checking the classification of SF and non-forested (NF) areas.

As a result, the GIS contained layers for two time periods, in which the vegetation and land use were separated into three classes, with their attribute tables, based on the classification by Ferraz *et al.* (2005), and described as follows:

Mature Forest (MF) – dense vegetation, with tall trees (> 20 m), with large stems and a continuous canopy. On the images, it was also observed the presence of luminosity inside the patches for definition of canopy openness.

Non-forests (NF) – areas with artificial/ man-made uses, such as sugarcane fields, other crops, villages, large equipments (industries) and paved grounds.

Secondary Forest (SF) – open secondary formations, with herbs, shrubs and/ or scattered arboreal plants, with many gaps and abundance of *Cecropia* trees, usually regenerating after abandonment of agriculture practices or due to selective logging.

Landscape description and data analysis

In order to analyze changes in forest cover and fragmentation, landscape and patch indices or metrics were calculated using data derived from GIS attribute tables for each time period. Metrics describing the landscape were (Ferraz *et al.* 2005; Armenteras *et al.* 2006; Echeverría *et al.* 2006): patch number, mature forest cover and isolation degree. For the latter, an average nearest neighbor index was calculated for the entire landscape using Euclidean distance for the two studied periods in the ArcGIS Spatial Statistics´ tool (ESRI 2006) followed by a comparison of observed *vs*. expected distribution and returning a distribution pattern ranging from clustered to dispersed. Additionally, an isolation index was calculated based on a point pattern analysis (Fortin and Dale 2005) adapted by Ribeiro *et al*. (2008). A total of 4,390 points were systematically distributed along the studied polygon, at every 250 m on x and y axis, for the two studied periods. At each point, the distance from the nearest fragment was obtained and the mean point distance calculated for the entire landscape composed the index.

Patch metrics used were: patch area and patch shape index or circularity index, defined as the ratio between the patch area and the area of a hypothetical circle with the same perimeter, assuming a value of 1 for areas perfectly circular (Meunier 1998; Martins *et al*. 2002; Nascimento *et al.* 2006).

All patches were assigned to three size categories: small (<30 ha), medium (30-200 ha) and large (>200 ha) fragments. Additionally, fragments were classified according to their shapes in "very irregular" (shape index < 0.4), "irregular" (shape index between 0.4 and 0.65) or "regular" (shape index > 0.65).

To calculate the average annual deforestation rate for MF area, the following formula was employed (Armenteras *et al*. 2006):

$$\text{Loss rate} = (\text{Ln}(A_{t1}) - \text{Ln}(A_{t0})) * 100)/ (t_1 - t_0)$$

where A = MF forest area (ha) at t_1 (2005) and t_0 (1975).

For year on year comparisons of number of fragments in size and shape classes a chi-square analysis (Zar 1996) was done. Paired t-tests for mean isolation and Kruskall-Wallis tests were carried out to compare means of metrics obtained between periods in the studied area.

RESULTS

Changes in forest cover

Between 1975 and 2005, there was a loss of MF cover and replacement by agricultural fields. Within the studied area, percentage of MF cover and number of fragments decreased from 45.61% (153 patches) to 24% (110 patches, of which 106 patches are presently owned by the São José Sugar Mill) (**Fig. 2**), indicating a forest area loss of 47.37% in 30 years, with an estimated annual loss rate of 2.14%. Landscape configuration remains the same since 2005, and according to local informants, it was defined at the beginning of the 1990 decade.

The largest patch existent in the studied area in 1975 was reduced to almost half of its extension, and the cleared forest areas were replaced principally by sugarcane fields.

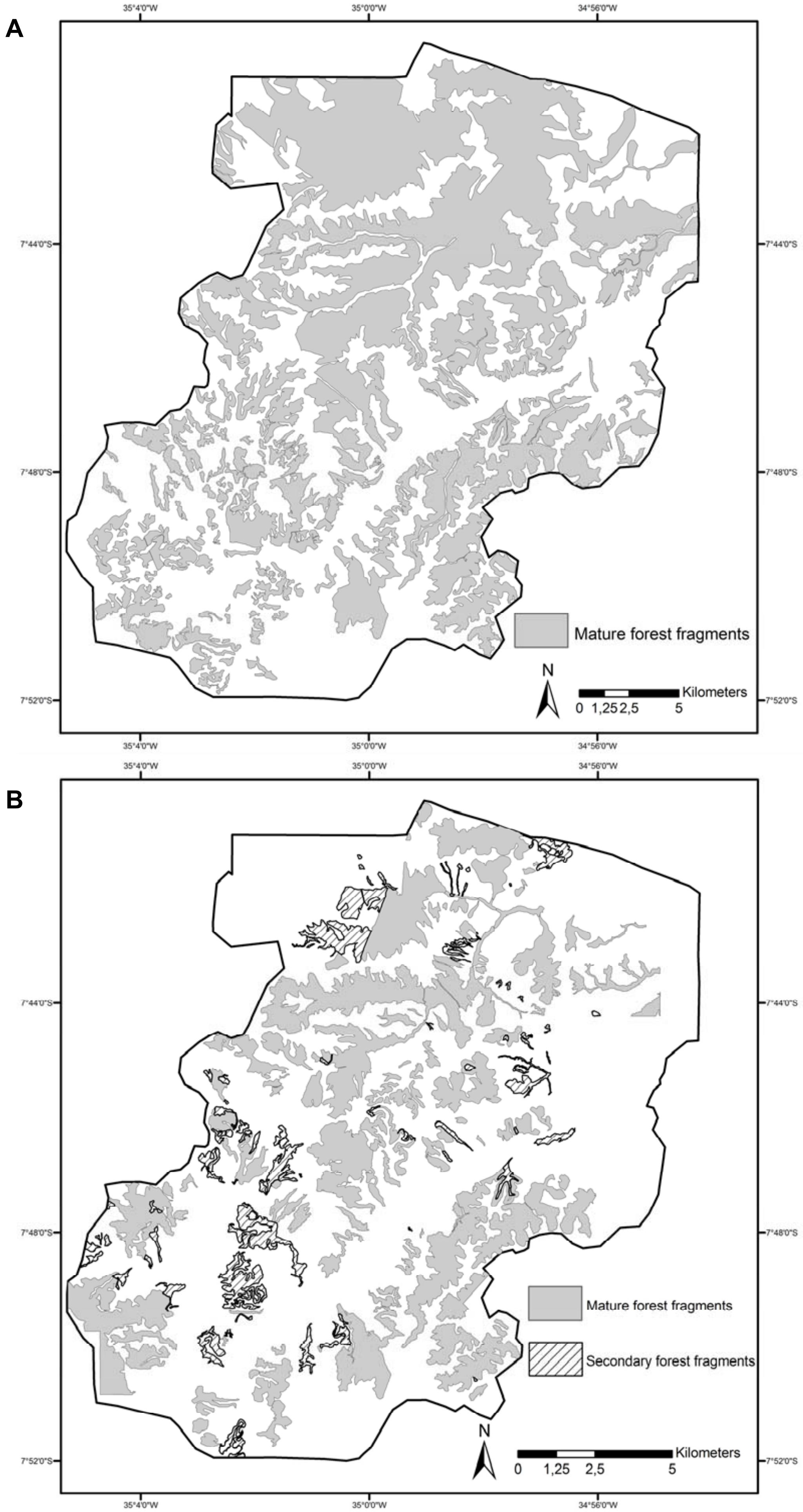

Fig. 2 Forest cover and spatial distribution of mature forests and non-forested areas in 1975 (A) and 2005 (B) in the northern region of the coast of Pernambuco, Brazil; secondary vegetation patches are shown in 2005.

This particular fragment was converted into 37 MF patches, with a mean size of 66.17 ha, and 31 open SF remnants, with a mean size of 11.44 ha, altogether covering 55% of its former area. An important feature in this process of fragmentation was the maintenance of corridors of riparian forests linking the remaining patches.

Variation in fragment size and shape

In 1975, small fragments were the majority (108), in addition to 33 medium and 12 large (**Fig. 3**). Inversely to numbers, all small fragments encompassed only 8.5% of MF areas, whereas the large ones covered altogether 70% of MF areas. In 2005, 70 small fragments were recorded, along with 29 medium and 11 large. The small fragments covered only 12.4% of the forested area, while large fragments, in a lesser number (10%), included 54.6% of the area and the medium patches (26%) covered 33% of total forest areas.

Although there has been a reduction in number in each size class after thirty years, the distribution of number of fragments per classes did not differ significantly between the years considered. Likewise, there were no significant differences considering the distribution of areas per size classes.

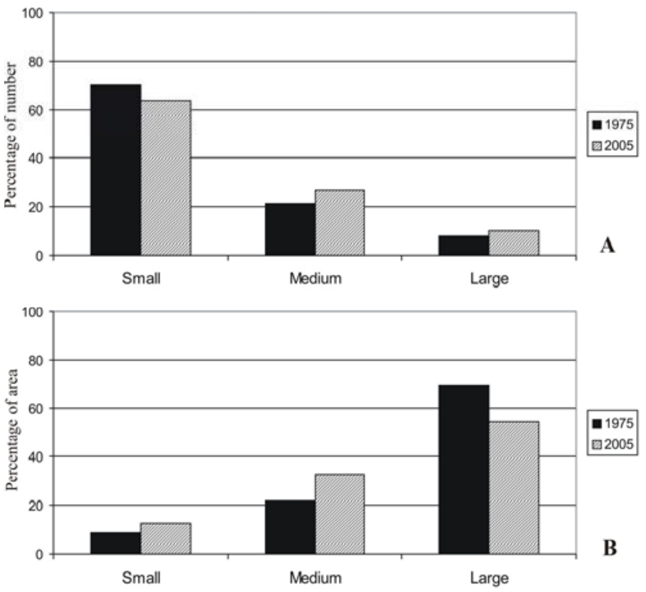

Fig. 3 Percentage of fragment number (A) and area (B) per size classes in 1975 and 2005, in the northern region of the coast of Pernambuco, Brazil.

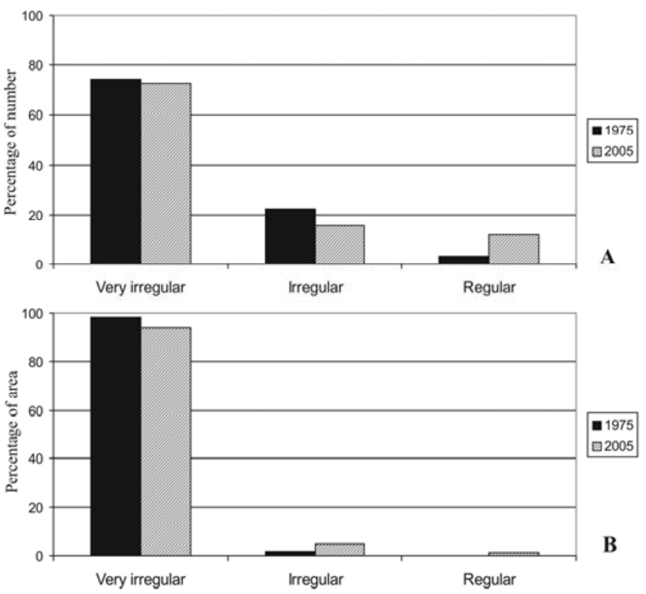

Fig. 4 Percentage of fragment number (A) and area (B) per shape classes in 1975 and 2005, in the northern region of the coast of Pernambuco, Brazil. The percentage of area occupied by regular fragments in 1975 was 0.08%.

Table 1 Landscape and patch indices for mature forests in 1975 and 2005 in the northern region of the coast of Pernambuco, Brazil

Landscape and patch indices	1975	2005
Patch number	153	110
Mature forest cover (%)	45.61	24.00
Average nearest neighbor (m)	397.10	695.97
Mean isolation index (m)	78.22	440.79
Mean patch area (ha)	83.47	61.10
Mean patch shape index	0.28	0.30

Mean patch area decreased from 83.47 ha in 1975 (minimum 1.14 ha, maximum 5090.88 ha) to 61.10 ha in 2005 (minimum 1.20, maximum 497.71 ha), with no significant differences for the means between the two studied years. Many small fragments were converted in non-forest areas. For instance, a total of 77 fragments in 1975 were entirely replaced by sugarcane fields and secondary forest, of which 68 were small and only two were large.

In 1975, almost all fragments in the study area had very irregular (74.5%) or irregular (22%) shapes, both classes covering together 99.9% of mature forest area, while only 0.1% of the area was occupied by regular patches (**Fig. 4**). All large and medium fragments and 64% of the small fragments had a very irregular shape; all regular fragments recorded were small ones.

In 2005, 72.7% of fragments had a very irregular or irregular (15.5%) shape, covering 98.7% of forest areas. All large and medium sized fragments, with the exception of two medium, and 60% of the small ones had very irregular shapes, and all regular patches were small, in the same way as in the former landscape.

Although the mean patch shape index did not differ between years (**Table 1**), the number of fragments per shape classes differed significantly over the period ($\chi^2 = 6.083$; df = 2; p = 0.04), principally because the proportion of very irregular and irregular decreased and regular fragments increased. This apparently positive result, however, should be seen with caution, because regular fragments comprise only 1.3% of actual MF area, and all of them have small sizes.

Forest isolation

In 1975, the observed fragment mean distance of nearest neighbor was 397.10 m, which, being compared to the expected mean distance of 602.96 m, revealed a clustered distribution (p = 0.01). In 2005, the observed nearest neighbor distance of was 695.97 m in average. Distribution pattern in the actual landscape, when compared to an expected average distance of 718.3 m, is then random, that is not clustered nor dispersed. Mean isolation indices calculated for the two periods showed a similar pattern of increasing isolation and reduction in the clustered pattern. The index changed from 78.22 m in 1975 to 440.79 m in 2005, with a significant difference between periods (t = 35.7, p < 0.01).

The isolation index indirectly incorporates another variable to the analysis, which is fragment area. In other words, if points fall within fragments, the distances from these points to surrounding fragments is zero. Therefore, in a landscape with large fragments, a great number of "zero" values would determine a lower isolation index. The number of points with "zero" values decreased from 2177 to 1070 m, whereas maximum point-fragment distance increased from 887 to 4557m. In fact, the scatterplot drawn with distances from nearest forests in all points in both years (**Fig. 5**) clearly shows that distances increased five times in 30 years. For instance, while there is a great concentration of points with distances between 25 and 500m in 1975, the majority of points concentrate distances from 200 to 2000 m in 2005 (**Fig. 5**).

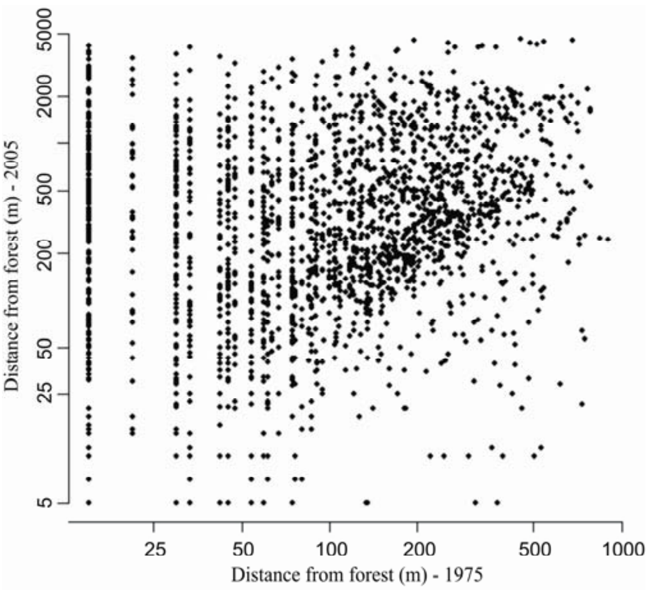

Fig. 5 Scatter plot showing distances of nearest fragment (in meters) for 4,390 points systematically distributed in the studied landscape, in 1975 *vs.* 2005, in the northern region of the coast of Pernambuco, Brazil.

Analysis of secondary vegetation

In the actual landscape, 96 patches of secondary forest (SF) were identified, with areas ranging from 0.12 to 130 ha and a mean patch area of 12.01 ± 23.29 ha), most of which classified as small (90.6%) (**Table 2**). All these small patches cover 44.3% of the area occupied by SF. The other 55.7% of SF areas were made up by medium-sized patches (9.4%), whereas no large patches of SF were recorded.

The shape of patches of SF vegetation in the studied area followed the same proportions recorded for MF patches: 79% very irregular (90.8% of total SF area), and 13% irregular (7.3% of SF area). In other words, this means that only 8% of the SF patches were regular, covering only 1.9% of SF areas. Amongst very irregular patches of SF, 89.5% are small; 91.6% of the irregular ones and all regular SF patches are small.

Comparing the two years, it was possible to identify that 76% of the SF patches existent in 2005 originated from MF areas. This conversion of MF into SF areas was frequent in patches with irregular or very irregular shapes, which are more sensitive to anthropogenic influence and management practices. Only 24% of the SF patches were originated from vegetation succession on abandoned farmed fields.

Table 2 Landscape and patch indices for secondary forest patches in 2005, in the northern region of the coast of Pernambuco, Brazil

Landscape and patch indices	2005 values
Patch number	96
Total Secondary forest cover (%)	4.12
Mean patch area (ha)	12.01
Patches adjacent to mature forests (%)	53.12
Mean patch shape index	0.29

DISCUSSION

Forest loss and fragmentation

This study represents the first historical analysis of landscape changes in the Atlantic Forest at the biogeographical sub-region Pernambuco, as defined by Silva and Casteleti (2005). Using high-resolution data, the analysis provides a precise quantification of deforestation, comparable to other assessments for the Brazilian Amazon (Ferraz *et al.* 2005; Armenteras *et al.* 2006). The studied landscape has undergone a high degree of deforestation, as showed by an overall annual loss rate of 2.14%. This rate is similar to the average deforestation rate in the Amazon in Rondônia, for two decades (2%; Ferraz *et al.* 2005), greater than the recorded for the western Amazon for 10 years (0.61%; Sierra 2000) or for most areas in the Colombian Amazon for 16 years (0.01-3.73%; Armenteras *et al.* 2006) and lower than calculated rates for endangered temperate forests in Chile (4.5%; Echeverría *et al.* 2006). We also found an actual mature forest cover of 24%, similar to the forest cover in the southern cost of Pernambuco (23%; Ranta *et al.* 1998) and in Southern Brazil (20.2%; Becker *et al.* 2004), but much lower than in the Amazon - 86% (Sierra 2000), 34% (Ferraz *et al.* 2005) and 28 to 99% (Armenteras *et al.* 2006) - and in the Atlantic Forest of São Paulo (32.03%; Silva *et al.* 2007).

Forest cover and loss found in this study precisely show that, although much of the fragmentation of the rain forest probably took place 300±500 years ago (Ranta *et al.* 1998), 45.71% of MF cover still existed in 1975. Hence, other 25% of total forest cover were cleared over the last three decades, very likely to have happened between 1975 and 1985, the 'próalcool' decade (Farias *et al.* 2007). This result is very different from the Amazon, which remained largely intact until the "modern" era of deforestation, begun with the inauguration of the Transamazon highway in 1970, after which deforestation proceeded at a rapid pace (Fearnside 2005). Considering that usually deforestation is not homogeneous in space (Echeverria *et al.* 2006) or time (Ferraz *et al.* 2005), a more detailed study using remote sensing data from the 1980 decade would possibly show variations in forest loss over time.

Overall, the fragmentation pattern in the studied area is very critical for forest conservation. Most fragments are small with irregular shapes and isolated. The degree of forest cover is below 30%, which is considered a critical threshold under which species loss occur very rapidly (Fahrig 2003), and even a small loss of habitat may result in a precipitous decline in the probability of metapopulation persistence (Ewers and Didham 2006). In general, differently from the ''fishbone'' patterns found in the Amazon, where landscape configuration is guided by roads (Ferraz *et al.* 2005; Armenteras *et al.* 2006), the configuration of remnants in the study area is apparently more related to relief, which has been found to be the main abiotic factor favoring the preservation of forest remnants in the Southeastern Atlantic Forest (Silva *et al.* 2007, 2008).

Along with forest loss, we reported a decrease in fragment number from 1975 to 2005. There are three scenarios that may explain this result: case 1) more negatively, a set of fragments were deforested and converted into sugarcane fields; case 2) fragments of mature forests were converted into secondary forest patches; and case 3) more optimistically, fragments were joined into larger patches. Our results revealed that each case occurred in the studied landscape.

Forest loss (case 1) was prevalent in the northern study area, where the level of modification was the highest, with fragmentation of a continuous forest into smaller patches. One of the major consequences of this severe fragmentation is the separation and isolation of populations. Habitat isolation is defined as the opposite of habitat connectivity or the degree of isolation between patches used by a particular species (*sensu* Fischer and Lindenmayer 2007). Indices of mean proximity are suitable for characterizing the degree of isolation and fragmentation in tropical forest landscapes (Echeverria *et al.* 2006) although they do not effectively take into account fragment area. In the study area, calculated average nearest neighbor increased from 1975 to 2005, the distribution pattern changed from clustered to random, and there was a reduction in mean fragment size. The point-based isolation index also revealed that, throughout the landscape, a great number of points which fell within forests in 1975 fell outside forest areas in 2005. Altogether, these results picture not only forest loss but a pattern of patch shrinking. Remaining patches becoming more and more isolated (Boentje and Blinnikov 2007) and a decline in the mean patch size is a general trend found in landscapes that progress towards a more fragmented structure (Çakir *et al*. 2008), with the main ecological driver of forest extent attributable to human activity (Wickham *et al*. 2007).

The effect of fragmentation and isolation on species is very difficult to evaluate, because dispersal ability largely varies among different groups of organisms. Habitat isolation affects species richness, as smaller and more isolated fragments are expected to retain fewer species (Debinski and Holt 2000). Moreover, isolation reduces the possibility of movements of organisms between fragments (Echeverria *et al*. 2006) and negatively affects the dispersal of species, creating patchy populations, whose interactions depend on dispersal behavior, mode and scale of movement (Fischer and Lindenmayer 2007). For instance, wind-pollinated trees can spread their pollen for tens of kilometers, whereas distance between fragments can be an insuperable barrier for some invertebrates, small mammals, large fruit-eating vertebrates (Forman 1995; Silva and Tabarelli 2000), sessile organisms with low dispersal power (Henle *et al*. 2004) and forest-interior species (Tabarelli and Gascon 2005). Isolation can discontinue genetic flow between populations in different fragments and consequently reduce genetic diversity (Kageyama and Gandara 1998; Silva EF *et al.* 2008).

Forest conversion (case 2) has occurred in a large scale as well, as 73 out of 153 MF patches were converted into SF fragments. SF patches derived from complete conversion of irregular shaped forest fragments or from degradation along fragment margins, where forest structure is more susceptible to edge effects (Murcia 1995). In the same area, it has been recorded that edge influence on vegetation floristics and structure is greater along the first 50 m from forest boundaries (Gomes *et al.* 2008; Silva HCH *et al.* 2008), although in Amazonian forests, edge effects can penetrate 500m or 2000m (Laurance 2000). Ranta *et al.* (1998) found out that in the southeastern corner of Pernambuco, the area influenced by the edge exceeded interior habitats, when edge width was greater than 60 m. Under edge influence, both physical and chemical environments are altered, affecting the distribution of species near the edge according to their physiological tolerances (Murcia 1995), leading to new ecological communities near habitat edges (Ries *et al.* 2004).

There is much debate about the conservation value of secondary patches *per se*. These secondary areas were found to be species-rich and important for the overall plant diversity (Castillo-Campos *et al.* 2008), and of lower value for small insects (Gardner *et al*. 2008). As a synthesis of these two view points, while conservation of forest remnants will not be possible by focusing solely on primary forests, there must be prudence with the claim that forest regeneration on degraded land can effectively compensate the loss of species following deforestation. In this study, medium and small patches of secondary forests are seen as potential areas for forest regeneration, with many of them adjacent, very close to or even linking single MF patches. These areas in the landscape can increase fragment areas, connect two or more patches or act as stepping stones, i.e., small areas of suitable habitats scattered within the matrix, facilitating biological flows between patches of suitable habitats (Metzger 2003b).

The joint of rainforest patches into larger ones (case 3) has also occurred in the study area. It was found that many large fragments in the present are resultant from a fusion of two to six small fragments in the past, particularly in the western portion of the area. Thus, the occurrence of case 3 shows that habitat loss and fragmentation can act at different paces, and that a reasonable amount of habitat connectivity (*sensu* Fischer and Lindenmayer 2007) can be restored with the regeneration of small areas.

Implications for conservation

For a successful conservation of forest remnants in the studied area, assuring the desired maintenance of biodiversity, increase of forested areas and the enhancement of habitat connectivity, this study points out for simple and effective actions, such as the union of small fragments to large fragments through reforestation, the creation of forest corridors following river courses, and the maintenance of patches of secondary vegetation.

Lessons for conservation priorities are derived from the spatial pattern and temporal changes analyzed in the study area, combined with conservation strategies defined for the Atlantic forest as a whole (Tabarelli and Gascon 2005). An important step, already achieved in the study area, is the establishment of partnership between research groups and the private sugar-alcohol sector for the execution of basic and applied biodiversity studies. According to Tabarelli and Roda (2005), the sugar-alcohol industry in Pernambuco has awakened to the need of producing more “environmentally correct” sugar, and protecting the water resources. The regional sugarcane sector has founded the Institute for the Preservation of the Atlantic Rainforest – (IPMA), which incentive the creation of private preserves, forest restoration and protection of the remaining forest. The São José Sugar Mill, landowner of the study area, is a member of IPMA and plays an important role in forest conservation and partnership, along with universities and research centers.

Secondly, there is an urgent need for the conservation of large fragments as forest reserves, as part of a regional strategy for the conservation of Atlantic forest in Northeastern Brazil, creating reserves to protect medium-to-large forest remnants (Silva and Tabarelli 2000), which would act as the source for propagules and seedlings, as indicated by reforestation models (MMA 2003). Although largest patches in the area are smaller than 500 ha, they are of great conservation value, mainly in such areas in which the percentage of forest left is very low (Turner and Corlett 1996).

Another important effective action is to enhance connectivity following river courses. The river network can act as an axis for the creation of ecological corridors, defined as linear and homogeneous areas in the landscape, linking at least two patches *sensu* Metzger (2003b). As part of the IPMA actions, there is an agreement to recover 6 ha of riparian forests per year, i.e. 1 km (60 m wide) of river margins reforested per year. Reforestation of riparian areas and of small areas linking forest patches may be an appropriate alternative. In simulating reforestation of sugarcane areas between fragments, Ranta *et al.* (1998) have shown an increase of 48% in forest cover and of 166% in forest core areas, given that edge habitats and fragment number would decrease considerably.

CONCLUSION

In the present study, major changes in landscape and forest cover in the Northeastern Atlantic forest of Brazil during the last thirty years have been quantified. Localized in the most heavily deforested part of the Atlantic forest dominion (Silva and Casteleti 2005), fragmentation pattern found over the last 30 years consists of a significant forest loss, associated with a decrease in patch area and increase in isolation, as well as decrease in patch number and shape. Although present forest cover represents a critical percentage for forest conservation (Ferraz *et al*. 2005), the result found is better than regional rates of Atlantic forest loss, which indicates a percentage of 1.5 to 4.6% (Lima 1998; Santos *et al*. 2001) of forest cover remaining in the State of Pernambuco. This signifies that, although Atlantic forest fragmentation is a conservation issue at a broad scale, mitigation actions require knowledge at local scale, due to peculiarities related to historical practices and local features.

Important consequences of landscape modification on biodiversity patterns and ecological processes have probably already taken place and many others are undergoing.

However, once the pattern of recent changes of a landscape is identified, conservation strategies can be drawn in order to mitigate biological consequences of such changes. Some actions towards a sustainable coexistence of sugarcane cultivation and forest conservation have already been started. For instance, clear-cut practices have stopped and are under policy, and there is an incentive for the creation of riparian forests enhancing connectivity. In this scenario, present forest cover must be maintained, and secondary forests play an important role, as they are seen as potential mature forests. From a conservationist and sustainable viewpoint, the actual scenario is then challenging and promising.

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REFERENCES

Armenteras D, Rudas G, Rodriguez N, Sua S, Romero M (2006) Patterns and causes of deforestation in the Colombian Amazon. *Ecological Indicators* **6**, 353-368

Becker FG, Irgang GV, Hasenack H,Vilella FS, Verani NF (2004) Land cover and conservation state of a region in the southern limit of the Atlantic forest (river Maquiné Basin, Rio Grande do Sul, Brazil). *Brazilian Journal of Biology* **64**, 569-582

Boentje JP, Blinnikov MS (2007) Post-Soviet forest fragmentation and loss in the Green Belt around Moscow, Russia (1991–2001): a remote sensing perspective. *Landscape and Urban Planning* **82**, 208-221

Çakır G, Sivrikaya F, Keles S (2008) Forest cover change and fragmentation using Landsat data in Maçka State Forest Enterprise in Turkey. *Environmental Monitoring and Assessment* **137**, 51-66

Câmara IG (2003) Brief history of conservation in the Atlantic Forest. In: Galindo-Leal C, Câmara IG (Eds) *The Atlantic Forest of South America: Biodiversity Status, Threats and Outlook*. Washington DC, Island Press, pp 31-42

Castillo-Campos G, HalVter G, Moreno CE (2008) Primary and secondary vegetation patches as contributors to Xoristic diversity in a tropical deciduous forest landscape. *Biodiversity and Conservation* **17**, 1701-1714

CPRH (Companhia Pernambucana do Meio Ambiente) (2003) *Diagnóstico Socioambiental do Litoral Norte de Pernambuco*. CPRH, Recife, 214 pp

Debinski DM, Holt RD (2000) A survey and overview of habitat fragmentation experiments. *Conservation Biology* **14**, 342-355

Ditt EH (2002) *Fragmentos florestais no Pontal do Paranapanema*. São Paulo, Annablume, 140 pp

Durigan G, Siqueira MF, Franco GADC (2007) Threats to the cerrado remnants of the state of São Paulo, Brazil. *Scientia Agricola* **64**, 355-363

Echeverría C, Coomesa D, Salasc J, Rey-Benayasd JM, Larab A, Newtone A (2006) Rapid deforestation and fragmentation of Chilean Temperate Forests. *Biological Conservation* **130**, 481-494

ESRI (Environmental Systems Research Institute) (2006) *ArcGIS Professional GIS for the desktop*, version 9.2. Redlands, ESRI

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

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

Farias GB, Alves AGC, Lins-e-Silva ACB (2007) Riqueza de aves em cinco fragmentos de Floresta Atlântica na Zona da Mata Norte de Pernambuco, Brasil. *Biotemas* **20**, 111-122

Fearnside PM (2005) Deforestation in Brazilian Amazonia: History, Rates, and Consequences. *Conservation Biology* **19**, 680-688

Ferraz SFB, Vettorazzi CA, Theobald DM, Ballester MVR (2005) Landscape dynamics of Amazonian deforestation between 1984 and 2002 in central Rondônia, Brazil: assessment and future scenarios. *Forest Ecology and Management* **204**, 67-83

Fischer J, Lindenmayer DB (2007) Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* **16**, 265-280

Forman RTT (1995) *Land Mosaics: The Ecology of Landscapes and Regions*, Cambridge University Press, Cambridge, 654 pp

Fortin MJ, Dale MRT (2005) *Spatial Analysis: A Guide for Ecologists*, Cambridge University Press, Cambridge, 380 pp

Gardner TA, Hernández MIM, Barlow J, Peres CA (2008) Understanding the biodiversity consequences of habitat change: the value of secondary and plantation forests for neotropical dung beetles. *Journal of Applied Ecology* **45**, 883-893

Goldemberg J, Coelho ST, Guardabassi P (2008) The sustainability of ethanol production from sugarcane. *Energy Policy* **36**, 2086-2097

Gomes JS, Lins-e-Silva ACB, Rodal MJN, Silva HCH (*in press*) Estrutura do sub-bosque lenhoso em ambientes de borda e interior de dois fragmentos de Floresta Atlântica em Igarassu, Pernambuco. *Rodriguésia* **59**

Henle K, Davies KF, Kleyer M, Margules C, Settele J (2004) Predictors of species sensitivity to fragmentation. *Biodiversity and Conservation* **13**, 207-251

Jorge LAB, Garcia GJ (1997) A study of habitat fragmentation in Southeastern Brazil using remote sensing and geographic information systems (GIS). *Forest Ecology and Management* **98**, 35-47

Kageyama PY, Gandara FB (1998) Conseqüências genéticas da fragmentação sobre populações de espécies arbóreas. *Série Técnica IPEF* **12**, 65-70

Köppen W (1936) Das Geographische System der Klimate. In: Köppen W, Geiger W (Eds) *Handbuch der Klimatologie*, Bd. I, Teil C, Kraus Verlag, Nendeln, pp 1-43

Laurance WF (2000) Do edge effects occur over large spatial scales? *Trends in Ecology and Evolution* **15**, 134-135

Lima MLFC (1998) *A Reserva da Biosfera da Mata Atlântica em Pernambuco - Situação Atual, Ações e Perspectivas*, São Paulo, Conselho Nacional da Reserva da Biosfera da Mata Atlântica, 45 pp

Martins ICM, Soares VP, Silva E, Brites RS (2002) Diagnóstico ambiental no contexto da paisagem de fragmentos florestais "ipucas" no município de Lagoa da Confusão, Tocantins. *Revista Árvore* **26**, 299-309

Metzger JP (2001) O que é ecologia de paisagens? *Biota Neotropica* **1**, 1-9

Metzger JP (2002) Landscape dynamics and equilibrium in areas of slash-and-burn agriculture with short and long fallow period (Bragantina region, NE Brazilian Amazon). *Landscape Ecology* **17**, 419-431

Metzger JP (2003a) Estrutura da paisagem: o uso adequado de métricas. In: Cullen Junior L, Pádua CV, Rudran R (Eds) *Métodos de Estudos em Biologia da Conservação e Manejo da Vida Silvestre*, UFPR/Fundação O Boticário de Proteção à Natureza, Curitiba, pp 423-453

Metzger JP (2003b) Delineamento de experimentos numa perspectiva de ecologia da paisagem. In: Cullen Junior L, Pádua CV, Rudran R (Eds) *Métodos de Estudos em Biologia da Conservação e Manejo da Vida Silvestre*, UFPR/ Fundação O Boticário de Proteção à Natureza, Curitiba, pp 539-553

Meunier IMJ (1998) Conservação da Reserva Ecológica de Dois Irmãos – Potencial e carências para a condução de um plano de manejo de área silvestre. In: Machado IC, Lopes AV, Pôrto KC (Eds) *Reserva Ecológica de Dois Irmãos: Estudos de um Remanescente de Mata Atlântica em Área Urbana*, Imprensa Universitária – UFPE, Recife, pp 291-307

MMA (2003) *Fragmentação de Ecossistemas: Causas, Efeitos Sobre a Biodiversidade e Recomendações de Políticas Públicas*, Ministério do Meio Ambiente (MMA), Brasília, 510 pp

Morellato LPC, Haddad CFP (2000) Introduction: The Brazilian Atlantic forest. *Biotropica* **32**, 786-792

Murcia C (1995) Edge effects in fragmented forest: implications for conservation. *Trends in Ecology and Evolution* **10**, 58-62

Nascimento HEM, Laurance WF (2006) Efeitos de área e de borda sobre a estrutura florestal em fragmentos de floresta de terra-firme após 13-17 anos de isolamento. *Acta Amazônica* **36**, 183-192

Nascimento MC, Soares VP, Ribeiro CAAS, Silva E (2006) Mapeamento dos fragmentos de vegetação florestal nativa da bacia hidrográfica do Rio Alegre, Espírito Santo, a partir de imagens do satélite IKONOS II. *Revista Árvore* **30**, 389-398

Radford JQ, Bennett AF, Cheers GJ (2005) Landscape-level thresholds of habitat cover for woodland-dependent birds. *Biological Conservation* **124**, 317-337

Ranta P, Blon T, Niemelã J, Joensuu E, Siitonen M (1998) The fragmented Atlantic rain Forest of Brazil: size, shape and distribution of Forest fragments. *Biodiversity and Conservation* **7**, 385-403

Ribeiro MC, Metzger JP, Martensen AC, Ponzoni FJ, Hirota MM (2008) The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. *Biological Conservation – Atlantic Forest Conservation Special Issue*, in press

Ries L, Fletcher RJJ, Battin J, Sisk TD (2004) Ecological responses to habitat edges: Mechanisms, models, and variability explained. *Annual Review of Ecology, Evolution and Systematics* **35**, 491-522

Saatchi S, Agosti D, Alger K, Delabie J, Musinsky J (2001) Examining fragmentation and loss of primary forest in the Southern Bahian Atlantic forest of Brazil with radar imagery. *Conservation Biology* **15**, 867-875

Santos JC, Uchôa Neto CAM, Carneiro RS, Albuquerque TCA (2001) *Diagnóstico das Reservas Ecológicas da Região Metropolitana do Recife*, Recife, Secretaria de Ciência e Tecnologia do Estado de Pernambuco, 45 pp

Sierra R (2000) Dynamics and patterns of deforestation in the western Amazon: the Napo deforestation front, 1986–1996. *Applied Geography* **20**, 1-16

Silva EF, Oliveira CAM, Lins-e-Silva ACB, Rodal MJN (2008) Diversity and genetic structure of natural fragmented populations of *Tapirira guianensis* Aubl. in Northeastern Brazil. In: Rodal MJN, Lins e Silva ACB, Gottsberger G (Eds) *The Fragments Project. Bioremediation, Biodiversity and Bioavaila-*

bility **2 (Special Issue 1)**, 35-40

Silva HCH, Lins-e-Silva ACB, Gomes JS, Rodal MJN (2008) The effect of internal and external edges on vegetation physiognomy and structure in a remnant of Atlantic lowland rainforest in Brazil. In: Rodal MJN, Lins e Silva ACB, Gottsberger G (Eds) *The Fragments Project. Bioremediation, Biodiversity and Bioavailability* **2 (Special Issue 1)**, 47-55

Silva JMC, Casteleti CHM (2005) Status of the Biodiversity of the Atlantic Forest of Brazil. In: Galindo-Leal C, Câmara IG (Eds) *The Atlantic Forest of South America: Biodiversity Status, Threats, and Outlook*, CABS & Island Press, Washington, pp 43-59

Silva JMC, Tabarelli M (2000) Tree species impoverishment and the future flora of the Atlantic Forest of northeast Brazil. *Nature* **404**, 72-74

Silva WG, Metzger JP, Bernacci LC, Catharino ELM, Durigan G, Simões S (2008) Relief influence on tree species richness in secondary forest fragments of Atlantic Forest, SE, Brazil. *Acta Botanica Brasilica* **22**, 589-598

Silva WG, Metzger JP, Simões S, Simonetti C (2007) Relief influence on the spatial distribution of the Atlantic forest cover on the Ibiúna Plateau, SP. *Brazilian Journal of Biology* **67**, 403-411

Tabarelli M, Gascon C (2005) Lessons from fragmentation research: improving management and policy guidelines for biodiversity conservation. *Conservation Biology* **19**, 734-739

Tabarelli M, Pinto LP, Silva JMC, Hirota M, Bedê L (2005) Challenges and opportunities for biodiversity conservation in the Brazilian Atlantic Forest. *Conservation Biology* **19**, 695-700

Tabarelli M, Roda SA (2005) An opportunity for the Pernambuco Endemism Center. *Natureza e Conservação* **3**, 128-134

Tanizaki K, Moulton TP (2000) A fragmentação da mata atlântica no estado do Rio de Janeiro e a perda de biodiversidade. In: Bergallo HG, Rocha CFD, Alves MAS, Sluys MV (Eds) *A Fauna Ameaçada de Extinção do Estado do Rio de Janeiro*, UERJ, Rio de Janeiro, 166 pp

Turner IM, Corlett RT (1996) The conservation value of small, isolated fragments of lowland tropical rain forest. *Trends in Ecology and Evolution* **11**, 330-333

Viana VM, Tabanez AAJ, Batista JLF (1997) Dynamics and restoration of forest fragments in the Brazilian Atlantic Moist Forest. In: Laurance WF, Bierregaard Jr. RO (Eds) *Tropical Forest Remnants: Ecology, Management, and Conservation of Fragmented Communities*, The University of Chicago Press, Chicago, pp 351-365

Wickham JD, Riitters KH, Wade TG, Coulston JW (2007) Temporal change in forest fragmentation at multiple scales. *Landscape Ecology* **22**, 481-489

Young A, Mitchell N (1994) Microclimate and vegetation edge effects in a fragmented podocarp-broadleaf forest in New Zealand. *Biological Conservation* **67**, 63-72

Zar JH (1996) *Biostatistical Analysis* (3rd Edn), Prentice Hall, New York, 662 pp