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# Forest cover, extinction thresholds and time lags in woody plants (Myrtaceae) in the Brazilian Atlantic Forest: resources for conservation

**Dary Moreira Gonçalves Rigueira · Pedro Luís Bernardo da Rocha · Eduardo Mariano-Neto**

**Abstract**   Efforts to conserve tropical forests could be strengthened based on ecological knowledge, such as extinction thresholds in ecological processes. Many studies of extinction thresholds associated with habitat reduction have focused on animals, generally at the patch scale. However, certain plant groups are very interesting models with which to study this type of relationship, such as Myrtaceae in Neotropical forests. Because trees are long-lived organisms, local extinctions in response to habitat loss may occur in different ways due to a time lag. In this study, our objective was to assess the occurrence of extinction thresholds at the landscape scale for Myrtaceae in a large biome and the pattern of species reduction in different tree size classes. We studied nine landscapes with different amounts of available habitat (between 5 and 55 % forest cover) in different parts of the Atlantic Forest in Bahia, Brazil, and in each landscape, we evaluated four plant classes based on tree circumference: saplings (CBH between 8 and 15 cm), young (CBH between 15 and 30 cm) adults (CBH $\geq$30 cm) and total (all individuals with CBH $\geq$8 cm). Landscapes with forest cover less than 25 % presented an approximately sixfold reduction in Myrtaceae total species richness compared with landscapes with forest cover greater than 40 %. We identified a relationship with a threshold between the amount of available habitat at the landscape level and Myrtaceae richness, with a reduction in total, sapling and

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young species below a threshold of 40 % forest cover, whereas adults had an extinction threshold at 30 % forest cover. We discuss the differences among the categories of plants associated with a time lag and the possibilities and limitations in applying these results in environmental management.

**Keywords**  Environmental management · Habitat loss · Local extinction · Long-lived organisms · Species loss · Tropical Forest

## Introduction

The use and occupancy of tropical areas by humans have been based on the replacement of forested landscapes with anthropogenic environments (Wright and Muller-Landau 2006; Tabarelli et al. 2010). This process has accelerated at an unprecedented rate over the last decades (Millennium Ecosystem Assessment 2005). Between 1,980 and 2,000, approximately 288 million hectares (21 % of the existing forests) were impacted (FAO 2001). The Brazilian Atlantic Forest biome harbors a disproportionate quantity of global biodiversity (Myers et al. 2000) and represents one of the most threatened forested biomes of the world; only 11.4–16 % of its original 150 million hectares remain, and more than 80 % of that is in patches of less than 50 hectares (Ribeiro et al. 2009). Nevertheless, the deforestation rate of this ecosystem remains high, estimated at 350 km$^2$ per year (Fundação SOS Mata Atlântica and INPE 2008).

This alarming habitat loss leads to both the reduction of biodiversity through local extinctions (Turner 1996; Fahrig 2001, 2003) and the loss of ecological processes and ecosystem services associated with the integrity of these environments and their biological populations. Estimates indicate that at least 15 ecosystem services, such as air quality regulation and natural hazards protection, have been degraded due to the unsustainable use of different global ecosystems (Millennium Ecosystem Assessment 2005). In response to the loss and fragmentation of natural habitats, efforts toward the conservation of tropical forests based on ecological knowledge have been promoted, such as the establishment of ecological corridors and ecological restoration initiatives (Bawa et al. 2004, Pardini et al. 2005, Rodrigues et al. 2009).

Ecological studies and theoretical models that provide assessment and understanding of the relationship between habitat reduction and species maintenance may provide a greater ability to predict local extinctions due to deforestation (Primack and Rodrigues 2001), such as extinction thresholds. The concept of thresholds is based on a nonlinear relationship between an ecological response (for example, the number of species) and habitat reduction, with a break point at a certain amount of available habitat (Toms and Lesperance 2003). The extinction threshold model can be regarded as a modification of the island biogeography model (MacArthur and Wilson 1967) but with different processes involved. The island biogeography theory proposes a relationship between area (islands) and the number of species based on resource availability, where larger areas (islands) are able to maintain a greater number of species and a lower extinction probability. Furthermore, islands closer to the mainland (source) could maintain greater richness due to the possibility of immigration (MacArthur and Wilson 1967). Unlike the species-area relationship, the extinction threshold model proposes a metapopulational relationship modulated by the amount of available habitat in the landscape through the interaction between habitat reduction and

structural changes in landscape geometry, with increased fragmentation and isolation; this interaction prevents connectivity among patches of natural habitat, reduces the viability of populations present in these environments, reduces the chances of recolonization, and favors local extinctions (Andrén 1994; Fahrig 2002, 2003).

Different levels of biological organization have been assessed in studies of extinction thresholds (Swift and Hannon 2010), but more studies have focused on populations than on biological communities. The occurrence of extinction thresholds in communities is still controversial in the literature. Some authors indicate that finding evidence for extinction thresholds in communities may not be feasible because the combination of specific thresholds for each species can linearize the relationship between the amount of available habitat and species richness (Hanski 1994). Another reason for the controversy is associated with the characteristics of different species: environmental disturbances may decrease the population of one group of species (specialists or forest species) but may favor increases of another group of species (generalists or open area species) in the same community, maintaining similar species richness values along the impact gradient (King and Baker 2011).

The hypothesis of an extinction threshold associated with habitat loss is primarily derived from computational models (Andrén 1994; Fahrig 2001, 2002; Flather and Bevers 2002). Several empirical studies to assess this relationship have been performed, especially in sub-tropical and temperate environments (Lindenmayer et al. 2005; Radford et al. 2005; Schmidt and Roland 2006; Montoya et al. 2010). Recently, Pardini et al. (2010) conducted a study of fragmentation thresholds with non-volant small mammals in the Atlantic Forest of São Paulo in southeastern Brazil. In this study, the authors noted the importance of the percent of available habitat (10, 30 and 50 %) and of patch size in the maintenance of specialist species (Pardini et al. 2010). However, studies conducted at the landscape level in tropical forests have primarily evaluated structural aspects of the landscape, such as patch size and connectivity (Martensen et al. 2008; Santos et al. 2008; Faria et al. 2009; Lopes et al. 2009), and few studies have assessed the influence of the amount of available habitat.

Most of these studies have focused on animal species (Blois et al. 2002; Swift and Hannon 2010). However, certain plant groups have certain characteristics that make them relevant to the evaluation of the threshold model, among them the plant family Myrtaceae. Myrtaceae is the fourth largest family of angiosperms in Brazil, represented by 23 genera and approximately 1,030 species (Giulietti et al. 2005; Souza and Lorenzi 2005). Common in rainforests and semi-deciduous forests in Brazil, Myrtaceae is one of the most important groups of woody plants in the Atlantic Forest biome (Souza and Lorenzi 2005); and particularly in this biome it is a typically forest-inhabiting, usually shade-tolerant group, generally dispersed by animals, that is frequent in the understory of mature forests and sensitive to anthropogenic change (Tabarelli and Peres 2002). Atlantic Forest is also an endemism center for several groups of Neotropical Myrtaceae (Stehmann et al. 2009). Their huge representation in the Atlantic Forest, either in abundance or richness, and their biological characteristics cause local richness of some genera, such a *Myrcia* sensu lato, to be a good indicator of the local diversity pattern in the biome (Murray-Smith et al. 2008). Thus, this plant group represents a suitable biological model to assess the extinction threshold in this landscape.

Adult individuals of tree species can remain in the landscape for a long time, even following land use transformation (Metzger et al. 2009), and local extinctions of these groups may occur generations after habitat loss and fragmentation in what is called a time lag in the biological response (Tilman et al. 1996; Kuussaari et al. 2009). However, as their

reproductive rates are drastically reduced, the effects of habitat loss and isolation on their reproductive and dispersal capacities are expected to be observed as a decrease or absence in the abundance of the youngest individuals (Tilman et al. 1996); thus, the threshold pattern is expected to be observed earlier among those groups more recently established on the landscape than among those groups with adults able to remain on the landscape for a very long time following habitat loss.

Empirically, a test of the extinction threshold model is not trivial and requires a continuous gradient of landscapes with different quantities of available habitat and relatively large habitat patches, enabling the inclusion of ecological processes relevant to the group; this requirement increases the total area of the study (Swift and Hannon 2010). By expanding the study area, various confounding variables that cannot be adequately controlled are incorporated, for example, different matrix types, habitat quality, historical factors, landscape structure, altitude, and latitude (Radford et al. 2005; Rhodes et al. 2008). Thus, the extinction threshold due to habitat loss should be sufficiently strong to be identified. The occurrence of a strong effect with an extinction threshold due to habitat loss could permit its use in predictive models between species richness and habitat area at the landscape scale, which is an easy quantification parameter for environmental management.

The Atlantic Forest is a large biome, represented by areas with different qualities and habitats with different types of land use and occupation, both at the national and regional levels. If the extinction threshold model is sufficiently strong to be identified at the landscape scale, with different confounding variables occurring (e.g., different habitat and matrix qualities), this could offer a simple and inexpensive method for decision making for management purposes through the measurement of vegetation cover in the landscape. Our objective in this study was to evaluate the influence of the amount of available habitat on the pattern of species loss of woody plants of the family Myrtaceae in the Atlantic Forest of Bahia, Brazil. Specifically, this study evaluated the following hypotheses: (i) Myrtaceae species richness responds to the amount of available habitat in the landscape; (ii) the decline in species number is non-linear, with an extinction threshold; and (iii) the pattern of species loss is different among sapling, young and adult plants.

## Methods

### Study area and selection of sample landscapes

The study was conducted in nine $6 \times 6$ km$^2$ landscapes, with each landscape having a different proportion of forest cover (between 5 and 55 %), representing the total amount of available habitat at the landscape level. The landscapes were sampled in the central and coastal portions of the Atlantic Forest biome, in the state of Bahia in northeastern Brazil (Fig. 1), a large region between latitudes $11°80'$ and $18°49'$ S and longitudes $21°24'$ and $40°08'$ W. This region has been characterized by an ancient process of use and occupation since the sixteenth century, primarily based on selective logging and the replacement of native vegetation with agriculture and livestock (Cavalcanti 2006). However, this process was further intensified in the twentieth century, especially in the mid-1940s through the 1960s, through the destruction of extensive forest remnants and commercial logging (Mendonça et al. 1994). The anthropogenic changes in the Atlantic Forest have shaped the current structure and quality of the forest remnants, promoting isolation and transforming most of these habitats into "islands" of secondary vegetation (Tabarelli et al. 2010).
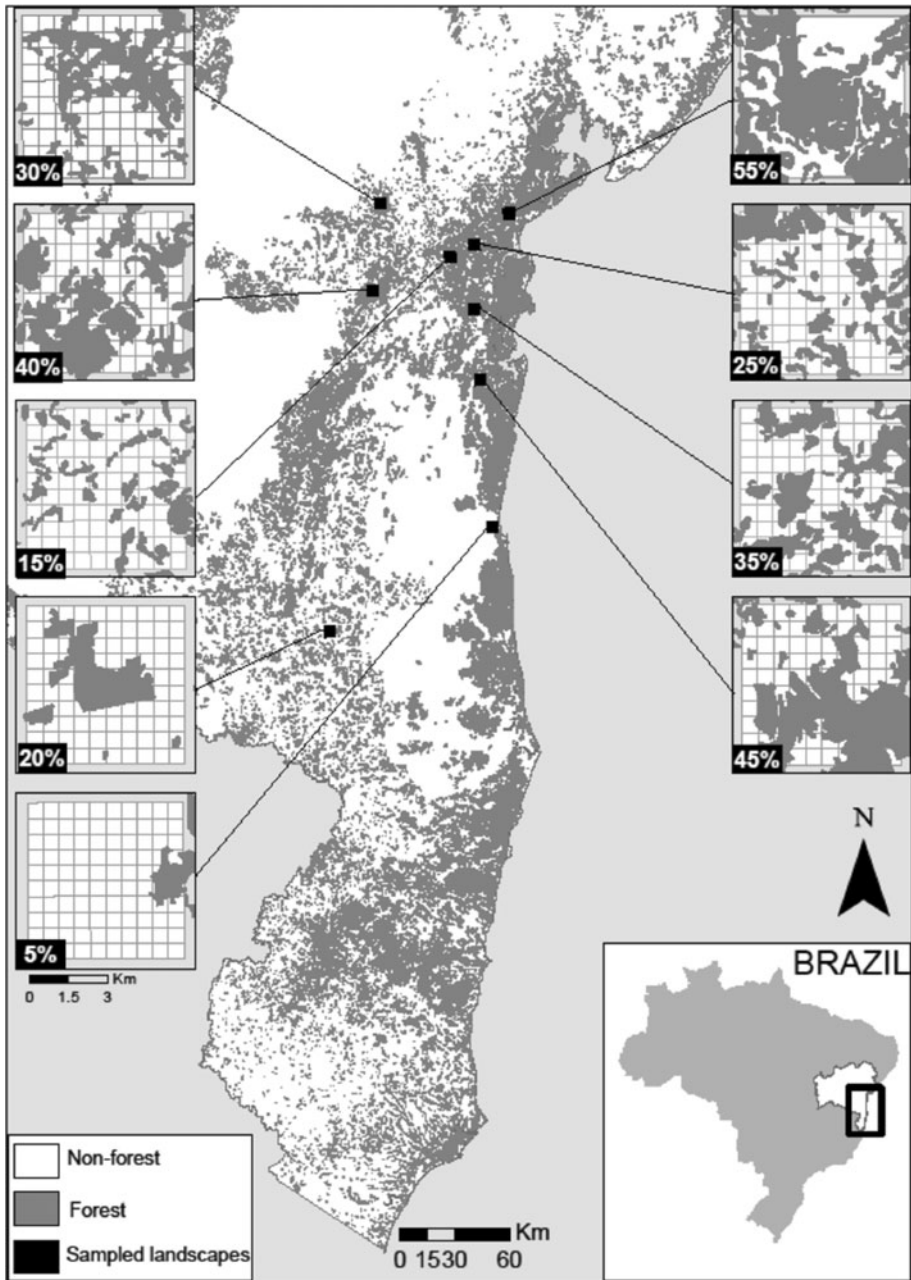
**Fig. 1** Map of the study area in the state of Bahia, Brazil, showing the remnants of Atlantic Forest (*gray areas*) and the nine sampled landscapes (*black squares*). In the details side, sampled landscapes and indicating their respective percentages of forest habitat. In detail at bottom right, map of Brazil indicating the limits of the state of Bahia (*white*) and the magnified region in the main part of the figure

The chosen landscape size, $6 \times 6$ km, appears to be suitable for including ecological processes relevant to tropical forest trees, such as the dispersal of their propagules and pollen, which are primarily restricted to relatively short distances (Condit et al. 1992; Gressler et al. 2006) The sampled landscapes were chosen from a distribution along two tracks of 1,500 6x6 km squares (36 km² or 3,600 ha) that contains most of the Atlantic Forest remnants in the state of Bahia. We used the forest cover map "Atlas dos Remanescentes Florestais da Mata Atlântica" (www.sosma.org.br and www.inpe.br), which was constructed through the manual classification of satellite images taken in 2008 (Landsar and CBERS). From this map, we calculated the proportion of forest cover in the $6 \times 6$ km landscapes. They were then classified according to the proportion of forest cover into classes of 5, 15, 20, 25, 30, 35, 40, 45 and 55 %. As theoretical and empirical studies rarely observe extinction thresholds greater than 55–60 % forest cover (Swift and Hannon 2010), we did not evaluate landscapes with higher proportions.

Before proceeding with the random selection of the landscapes, we established selection criteria to minimize the effects of four relevant variables that can be controlled via design: two related to the forest quality (successional stage) and matrix (degree of permeability to typical forest species) and two related to the quantitative indicators of forested habitat in the surrounding landscape (habitat amount and presence of potential source areas).

Because richness and composition vary among forests in different successional stages (Santos et al. 2008), we decided to include in the sample only landscapes with forests in intermediate or advanced stages of succession, as proposed in the CONAMA Resolution 05/1994, from a Brazilian council of the environment (CONAMA 1994); by these definitions, we sampled only forest tracks with a mean canopy height of at least 8 m and only rare presence of early successional plant groups.

As matrices (non-habitat) that are structurally similar to forests are more permeable to typically forest organisms and can minimize the effects of habitat loss and fragmentation in the landscape (Ewers and Didham 2006), we established in this study that the landscape matrices should be composed of at least 80 % low-profile, anthropogenic vegetation, such as pasture fields or upland crops/shrubs, and we excluded urban areas.

To avoid any possible effect of the surrounding area on the sampled landscape ($6 \times 6$ km) due to a larger area of forest and/or larger fragments (serving as a source area), for each $6 \times 6$ km landscape, we established an $18 \times 18$ km square centered on the landscape, and we used it to compute the percentage of vegetation cover and the area occupied by the largest patch (Largest Patch Index—LPI—Fragstats 3.3). We excluded landscapes for which the percentage of habitat cover and the LPI in the $18 \times 18$ km square were greater than in the $6 \times 6$ km sampled landscape.

The landscapes that we analyzed were Ilhéus—5.10 % (class 5 %), Presidente Tancredo Neves—15.65 % (class 15 %), Itapetinga—20.02 % (class 20 %), Valença—24.39 % (class 25 %), Ubaíra—30.14 % (class 30 %), Nilo Peçanha—33.11 % (class 35 %), Wenceslau Guimarães—40.14 % (class 40 %), Camamú—45.01 % (class 45 %), and Jaguaripe—54.15 % (class 55 %).

After defining the sampled landscapes, we evaluated other variables (environmental and landscape) that could influence richness in different landscapes but that we cannot control in sampling design because it would not be feasible to select sufficient landscapes for this study. These environmental and landscape variables were altitude, precipitation, proportion of edge and core areas in the landscape (defining edge areas as 50 m from any boundary). We also tested the spatial autocorrelation using a Mantel test between a geographical distance matrix and a matrix of differences in species richness between pairs of landscapes.

Sample plots and collection of biological data

Myrtaceae richness in each landscape was evaluated for a set of eight $10 \times 25$ m plots that were randomly located inside each landscape, at a minimum distance of 50 m from any forest edges. To locate each plot, we overlaid a grid of 100 cells ($600 \times 600$ m each) on the landscape and randomly selected eight cells that were at least partially covered with forest. In each sampled plot, we also evaluated successional stage and estimated the canopy height and the height of emergent trees using a 12 m pole.

We sampled circumference and estimated heights of all Myrtaceae with CBH $\geq 8$ cm. Branches were collected and returned to the FLORA laboratory (IBio/UFBA) for processing. The Myrtaceae species were identified by specialist Marcos Eduardo G. Sobral from the Federal University of São João del Rei (UFSJ) in Minas Gerais, Brazil and stored in the collection of the Alexandre Leal Costa Herbarium of the Federal University of Bahia (ALCB/IBio/UFBA). The collection of the botanical material was performed under the authorization of collection 12023-1 IBAMA/ICMBio.

The plants were divided into four circumference classes that allowed inference regarding groups of individuals that were established in different time periods in the landscape: class 1—total richness (individuals with CBH $\geq 8$ cm); class 2—saplings (individuals with CBH between 8 and 15 cm); class 3—young (individuals with CBH between 15 and 30 cm); and class 4—adults (individuals with CBH $\geq 30$ cm).

Data analysis

The possible influences of uncontrolled environmental and landscape variables in this study (altitude, rainfall, spatial correlation and the proportion of core and edge areas in the landscape) on Myrtaceae species richness were evaluated using simple linear regression.

To evaluate the existence of a threshold in the relationship between forest cover and Myrtaceae richness, we compared models with and without thresholds. We used piecewise regression models and generalized linear models (GLM), and we compared those models via Akaike information criterion (AIC), we accepted as the most probable model the one with the lowest AIC value and $p < 0.05$. Models with AIC values with a difference of less than two units were considered equally probable. Piecewise regression models are based on fitting to the data one or more lines united by one or more "breakpoints" representing the thresholds (Toms and Lesperance 2003). Piecewise models are considered suitable for ecological studies that evaluate nonlinear relationships, such as thresholds, and are commonly used in those cases (Toms and Lesperance 2003; Ficetola and Denoël 2009; Swift and Hannon 2010). In all models, we assumed a Poisson error distribution for the species richness data. The significance of the threshold values was calculated using Davies tests, based on the significance of the difference between the slopes of the fitted lines (Muggeo 2008).

To assure that the pattern found was not an artifact, we also build richness estimator curves and calculate the expected richness for all areas, and we tested the effects of forest cover over the estimated richness.

All analyses were performed in R 2.15 (R Development Core Team 2012) using the packages "segmented" 0.2–8.4 (Muggeo 2012) for the piecewise regression, "vegan" for the richness estimators curve, and "stats" for the GLM models.

## Results

Characterization of the sampled landscapes

Some landscapes had a predominance of pastures in the non-forested matrix (in the landscapes with 30 and 20 % forest cover), whereas in others, small-scale or family agriculture (40 and 25 %) or a mosaic of farmland with pastures (55, 45, 35, 15 and 5 %) were common (Appendix Table 1).

We found no correlation between mean altitude and forest cover ($p = 0.899$) or Myrtaceae richness. The annual average rainfall varied from 946 mm (20 %) to 2,572 mm (45 %) (Appendix Table 1) and was not correlated with the available habitat amount ($p = 0.507$). No spatial correlation was observed between the sampled landscapes and richness ($p = 0.323$).

The largest patches were observed in the landscapes with greater forest cover (55 and 45 %); however, large patches were also observed in the landscapes with less forest cover (30 and 20 %) (Appendix Table 1). We observed a significant decrease in maximum patch size with decreasing forest cover in the landscape ($p = 0.009$). However, there was no significant relationship between forest cover and the proportion of edge areas ($p = 0.063$) or the proportion of core areas ($p = 0.251$).

Myrtaceae richness

We collected 470 individuals of Myrtaceae representing 11 genera and 174 species, including 268 individuals (57.02 %) with CBH between 8 and 15 cm (sapling), 134 (28.51 %) with CBH between 15 and 30 cm (young), and 68 (14.47 %) with CBH greater than 30 cm (adult). The two most representative genera for this family contributed 63.2 % of the total richness: *Eugenia* with 75 species and *Myrcia* with 35 species (Appendix Table 2).

The highest values of richness for the sapling, young and total classes were obtained in the three landscapes with the greatest forest cover: 55, 45 and 40 % forest cover with 40, 32 and 39 species (sapling), 13, 18 and 24 species (young) and 47, 43 and 57 species (total), respectively. Among the adults, we found the highest richness in the landscapes with 30 and 40 % forest cover (12 and 14 species) and otherwise similar values in the other landscapes with greater than 25 % (between 6 and 7 species). For the landscapes with less than 20 % forest cover, we found lower values: 2, 3 and 1 species (Appendix Table 1).

Richness estimator curves

Rarefaction curves by landscape showed that the landscapes with lower cover areas, between 5 and 25 %, tends to reach an asymptote and also presented lower confidence intervals. We did not attain an stabilization in landscapes with higher vegetation cover, which also exhibits larger confidence intervals in rarefactions curves (Fig. 2). This results probably indicates that we have a good survey on landscapes with vegetation cover lower than 25 %, and we can expect even higher richness on landscapes above 40 %.

Threshold evaluation—piecewise regression

The piecewise models were the most probable (lower AIC and $p < 0.05$) models in all analyses. We identified breakpoints in the relationship between forest cover and species
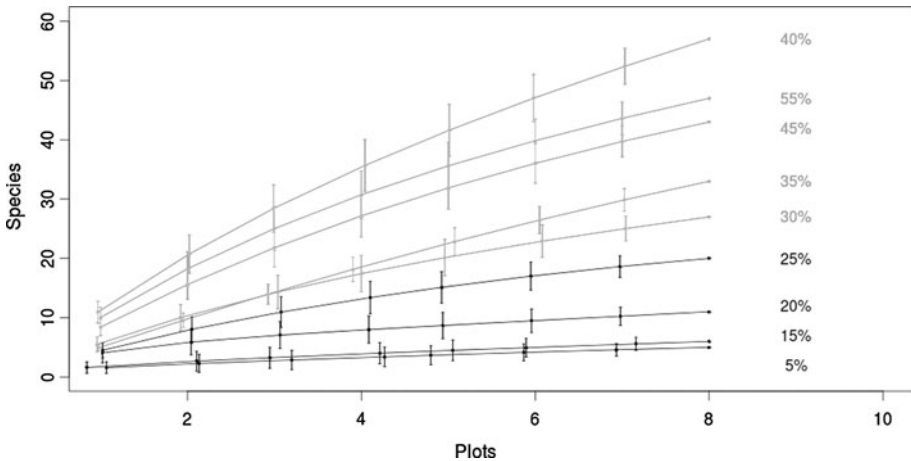
**Fig. 2** Rarefaction curves of the each landscape for Myrtaceae total richness. The percentage values in the right side represents the amount of available habitat in each landscapes

richness in all analyzed classes but with different values (Fig. 3). For total richness, the breakpoint was $40.0 \pm 2.6$ % forest cover ($p < 0.000$); for saplings, $40.1 \pm 5.1$ % ($p = 0.04$); for young, $40.0 \pm 4.1$ % ($p = 0.004$); and for adults, $30.0 \pm 4.1$ % ($p = 0.009$). The models using estimated total richness against forest cover showed the same response as original total richness data (Fig. 4), with breakpoint at $40.0 \pm 2.6$ % forest cover ($p < 0.000$).

## Discussion

### Forest cover and species richness

From these results, we conclude that Myrtaceae species richness was greatly reduced in landscapes with less forest cover. When including the standard deviations, we found that there is a high probability of a threshold existing between 25 and 45 % forest cover at a landscape level for all classes. Comparing the average Myrtaceae richness between a group of three landscapes with greater forest cover (55, 45 and 40 %) and a group with less forest cover (20, 15 and 5 %), we observed a 6.7-fold difference in total richness. We also did not observe a significant relationship with any environmental variable, such as rainfall, altitude, spatial correlation or core/edge proportion, but forest cover at the landscape level seems to be a good predictor of Myrtaceae richness in the landscape. This study represents an important empirical advance in knowledge regarding the extinction threshold theory associated with habitat loss in tropical environments and is a pioneer work for Myrtaceae and woody plants in tropical forests.

These results are consistent with those of Montoya et al. (2010), which showed that a decrease in the amount of available habitat was correlated with a decrease in the probabilities of tree species occurrence in Spain and with an increase in eventual species extinctions in fragmented landscapes; however, these authors argued that the effects of habitat fragmentation on tree species are weak relative to the effects of the amount of available habitat. Empirical studies conducted with different taxa, such as fungi (Drinnan 2005), insects
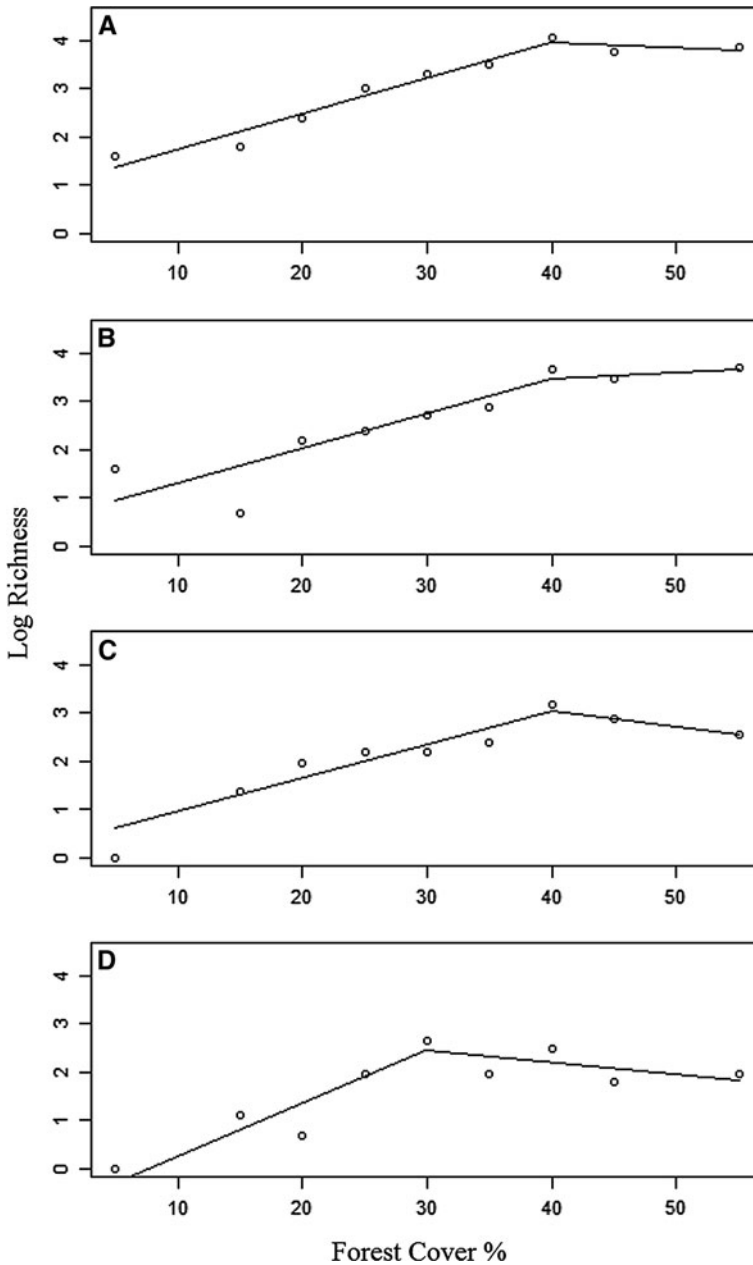
**Fig. 3** Piecewise regression model fit showing relation between forest cover and Myrtaceae richness. **a** Total richness, **b** sapling richness, **c** young richness, **d** adult richness. Richness values of the piecewise regression model were logarithm

(Schmidt and Roland 2006), birds and lizards (Lindenmayer et al. 2005) and small mammals (Pardini et al. 2010), also showed a direct relationship between the amount of available habitat and species richness.
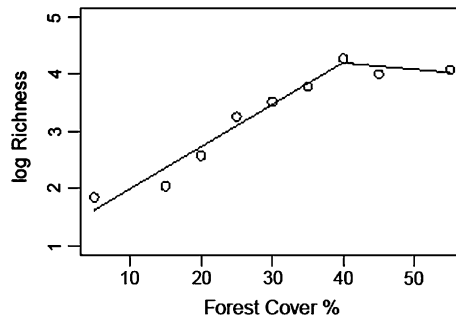
**Fig. 4** Piecewise regression model fit showing relation between forest cover and Myrtaceae estimated total richness. Estimated total richness value of the piecewise regression model was logarithm

The results also indicate that even human-altered landscapes with an intermediate level of forest cover in the landscape (e.g., landscapes with 35–55 % forest cover) maintain many more species than landscapes with lower forest cover (e.g., 5–20 %). As the components of ecosystems interact and contribute to maintaining ecosystem functionality and sustainability (Bennett et al. 2009), the decrease in the number of Myrtaceae species, one of the most important families of woody plants in the Atlantic Forest (Souza and Lorenzi 2005; Stehmann et al. 2009), may represent a reduction in the availability of important resources, especially for birds, monkeys, and bees, the primary dispersers and pollinators for this family in Brazil (Tabarelli and Peres 2002; Tabarelli et al. 2004; Gressler et al. 2006; Oliveira et al. 2010), which may influence the maintenance of different biological groups and ecological processes in these landscapes.

By doing rarefaction curves to assess whether the sampling effort conducted could encompass an approximate number of species expected in each landscape, we observed that the landscapes between 5 and 25 % of the habitat reached a stable platform, on the other hand the landscapes with a greater amount of habitat did not stabilized. These results reinforcing our view that the scenario of species loss that we found may have been conservative in terms of breakpoint value or abrupt change in richness.

Studies that seek to evaluate the occurrence of extinction thresholds primarily use species abundance (population level) as the response variable, with a focus on animal groups (Swift and Hannon 2010). Some authors suggest that the occurrence of thresholds for values of species richness in communities is unlikely (Hanski 1994; King and Baker 2011). As biological communities are generally composed of species that exhibit different responses to environmental disturbances, the most sensitive species tend to be replaced by less sensitive species but at different replacement rates; thus, no relationship between environmental diversity and species richness would be observed (King and Baker 2011).

The family Myrtaceae in the Atlantic Forest consists of primarily forest species that are the major component of understory surveys, typical understory trees, usually shade tolerant, generally dispersed by animals and sensitive to environmental changes (Silva and Leitão-Filho 1982; Mori et al. 1983; Melo and Mantovani 1994; Guedes-Bruni et al. 1997; Silva and Nascimento 2001; Tabarelli and Peres 2002). In a recent study, Montoya et al. (2010) evaluated the occurrence of extinction thresholds for different tree species with different biological characteristics. Of the eight studied species, only one (*Fagus sylvatica*) showed an extinction threshold, with this species being strictly a forest species, more shade tolerant and sensitive to anthropogenic changes than the other species evaluated (Montoya

et al. 2010). Possibly the biological characteristics of the Myrtaceae as a whole in the Atlantic Forest, allow this group to present such response to the amount of available habitat. Considering that normally this group is one of the most representative woody plant of the Atlantic Forests, our results indicate that the species of this family have substantially reduced their representation in landscapes with small percentage of forest cover.

## Habitat loss and time lag

We found different threshold values between the adult class and the other classes, and this difference is most likely associated with the time lag of these groups facing habitat loss. Trees can have time lag responses as some species have estimated life spans exceeding 100 years (Paltto et al. 2006). As the processes of habitat loss and fragmentation in the Atlantic Forest of Bahia were more intensified in the last 40–60 years (Mendonça et al. 1994), it is possible that some of the present adults witnessed the process of deforestation. These organisms with great longevity are capable of remaining on the landscape even after it changes, responding slowly to this change (Metzger et al. 2009).

However, their reproductive rates are drastically reduced, as is their ability to disperse and to recolonize other fragments in a severely fragmented landscape to maintain viable populations (Tilman et al. 1996). In this way, the structure and richness of saplings and young plants tends to reflect the current quality and configuration of the landscape.

Thus, different explanations are required for the observed patterns of species reduction and the extinction thresholds among the size classes in this study. Most likely, the reduction in the number of adults is associated with the non-replacement of this class, representing a gap in the recruitment of new individuals; thus, adults remain in landscapes even with low amounts of available habitat, and they have lower threshold values than the other classes. Unlike the adult plants, the loss of saplings and young trees occurs early in the land use gradient and is most likely related to decreased recruitment and the inability to recolonize the landscape, indicating the real sensitivity of Myrtaceae species to habitat loss. The absence of regenerating plants today will mean the absence of adult plants in the future.

This higher sensitivity of saplings, especially of shade-tolerant species such as the Myrtaceae plants in the Atlantic Forest, has been observed in another study also conducted in the Atlantic Forest (Pardini et al. 2009). Another typical forest family, Sapotaceae, was studied with the same sample design as in this study (Lima and Mariano-Neto, unpublished data). This family has some characteristics similar to Myrtaceae, but it is composed primarily of large trees. In that study, the plants were not divided into classes, and an extinction threshold was found at 30 % habitat cover for the entire assemblage, with a severe reduction in richness, as observed in Myrtaceae in this study. The lower threshold value is most likely associated with a time lag because the individuals were not divided into size classes.

## Resources for conservation

The existence of extinction thresholds associated with habitat loss is a potential tool for environmental management (Huggett 2005; Groffman et al. 2006; Rigueira 2011). Having identified a threshold, it is possible to propose appropriate management of the landscape to maintain or restore forest cover values above that threshold, which is more likely to retain a

greater proportion of the species richness (Lindenmayer et al. 2005; Radford et al. 2005). Despite this, the use of knowledge regarding extinction thresholds in conservation plans has been criticized by some authors who emphasize the variability of the threshold values obtained in different studies (Huggett 2005; Lindenmayer and Luck 2005). In an empirical study conducted in Australia, the same species of marsupial was observed to have different threshold values depending on the landscape studied, which indicated that such variation might have occurred due to differences in habitat quality, demographic rates and different types of land use and land cover on the landscape (Rhodes et al. 2008).

However, in this study, we observed consistent evidence that the entire Myrtaceae assemblages were strongly reduced when forest cover was less than 25–30 %; most likely, 35–40 % forest habitat, whether natural or associated with agroforestry systems, is necessary to maintain the great Myrtaceae diversity in the Atlantic Forest as well as in other forest types. Therefore, the use of an easily obtained landscape metric, such as the amount of available habitat, can be a good indication of the minimal forest proportion that must be maintained in the landscape, ensuring high richness of major woody plant groups (and their associated ecological processes) in the Atlantic Forest.

These results reveal a critical scenario for the Myrtaceae community of the Atlantic Forest, especially following the recent revision of the principal Brazilian environmental law: the Forest Code (Law 4.771/1965). In Brazil, all rural properties were divided into three groups: agricultural land, legal reserves (LR) (natural habitat areas subjected to sustainable management, with no clear cutting), and the areas close to rivers, dams, high slopes and sensitive areas that are designated as permanent preservation areas (PPA). If this law were enacted in Brazil, it is estimated that PPA would occupy 10 % of the total area, that LR would be 20 % of the property in the Atlantic Forest, and that the total forested areas would be approximately 30 % in all landscapes. Following other modifications to the law (Law 12.651/2012), the forested areas were estimated to decrease to include only the 20 % legally preserved habitat among the rural properties in the Atlantic Forest biome. In addition, maintaining LR for small rural properties is no longer required, which may represent a major loss of forests in those territories of this biome that are composed primarily of this type of property. Thus, the conservation goals of the new Forest Code would be very inefficient, considering that the minimal forest cover of 20 % represents a value that we found led to a severe reduction in Myrtaceae richness and considering that 20 % is below the threshold observed for sapling, young and adult species.

Our results showed that, on average, landscapes with high amounts of available habitat (greater than 40 %) had greater than four times the number of species found on a landscape with 20 % forest cover. Based on the precautionary principle, which is an important principle for making environmental decisions given the uncertainties and limitations of ecological data, a preventive margin related to potential and plausible threats should be ensured (United Nations 1992). Based on the results of this study, it appears that protecting only 20 % of the forest cover in the Atlantic Forest is not a preventive margin. For the long-term conservation of biodiversity and ecosystem services, which depend upon both the protection of natural habitats and on maintaining the integrity of the ecological processes that sustain them, a conservation strategy that is based on minimal amounts of available habitat for a representative group of the Atlantic Forest such as Myrtaceae may

well ensure the conservation of multiples species, maintaining the important ecological processes derived from the interactions among different species at the landscape scale.

Landscape metrics used, such as the amount of available habitat, for decision making in environmental management, identifying priority areas for conservation and restoration can be determined in a less costly way, based on satellite images (Banks-Leite et al. 2011). This is, however, only one potential tool to be used by environmental managers. Integrating different ecological concepts into a multi-scale perspective is crucial for the successful conservation of the Atlantic Forest. As an example, Rigueira et al. (2013) propose a hierarchical analysis based on three different spatial scales (regional, landscape and local) based on the theories of ecological thresholds and the principles of landscape ecology, ecological succession and the precautionary principle to evaluate the environmental feasibility of requests for vegetation removal in the state of Bahia, Brazil.

## Appendix

See Tables 1 and 2.

**Table 1** Table characterization of the landscapes and plots sampled in this study

| Landscape | Coordinates | Vegetation | Altitude (m) | Precipitation (mm) | Matrix (non habitat) | LPI/LPA | Landscape edge proportion | Myrtaceae richness (species) | Plot | Successional stage | Canopy height (m) | Emergent trees height (m) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Ilhéus (5 %) | 39°06′00′′W 14°44′10′′S | Dense rainforest | 92 | 2,069 | Mixed: presence of pastures, agriculture and industrial hub | 5.02 % (180 ha) | 28.80 % | Total (5) Class 2 (5) Class 3 (1) Class 4 (1) | 1 | Intermediate | 10–12 | 20 |
| | | | | | | | | | 2 | Advanced | 11–14 | 30 |
| | | | | | | | | | 3 | Intermediate | 10–12 | 20 |
| | | | | | | | | | 4 | Intermediate | 10–12 | 22 |
| | | | | | | | | | 5 | Intermediate | 10–12 | 18 |
| | | | | | | | | | 6 | Intermediate | 11–14 | 22 |
| | | | | | | | | | 7 | Advanced | 11–14 | 23 |
| | | | | | | | | | 8 | Intermediate | 11–14 | 17 |
| P.Tancredo Neves (15 %) | 39°18′57′′W 13°23′24′′S | Dense rainforest | 185 | 1,557 | Mixed: presence of pastures and small-scale agriculture | 2.2 % (79 ha) | 52.60 % | Total (6) Class 2 (2) Class 3 (4) Class 4 (3) | 1 | Intermediate | 8–10 | 17 |
| | | | | | | | | | 2 | Intermediate | 9–11 | 20 |
| | | | | | | | | | 3 | Intermediate | 6–8 | 15 |
| | | | | | | | | | 4 | Intermediate | 8–10 | 18 |
| | | | | | | | | | 5 | Advanced | 10–13 | 22 |
| | | | | | | | | | 6 | Intermediate | 8–12 | 20 |
| | | | | | | | | | 7 | Advanced | 10–14 | 22 |
| | | | | | | | | | 8 | Intermediate | 6–8 | 13 |
| Itapetinga (20 %) | 39°56′21′′W 15°15′12′′S | Semideciduous forest | 262 | 946 | Pastures | 18.27 % (648 ha) | 16.53 % | Total (11) Class 2 (9) Class 3 (7) Class 4 (2) | 1 | Advanced | 10–12 | 25 |
| | | | | | | | | | 2 | Intermediate | 10–12 | 25 |
| | | | | | | | | | 3 | Advanced | 11–15 | 25 |
| | | | | | | | | | 4 | Intermediate | 10–12 | 25 |
| | | | | | | | | | 5 | Intermediate | 8–10 | 20 |
| | | | | | | | | | 6 | Advanced | 9–12 | 22 |
| | | | | | | | | | 7 | Intermediate | 10–13 | 30 |
| | | | | | | | | | 8 | Advanced | 11–14 | 30 |

**Table 1** continued

| Landscape | Coordinates | Vegetation | Altitude (m) | Precipitation (mm) | Matrix (non habitat) | LPI/LPA | Landscape edge proportion | Myrtaceae richness (species) | Plot | Successional stage | Canopy height (m) | Emergent trees height (m) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Valença (25 %) | 39°11′27″W 13°19′51″S | Dense rainforest | 146 | 2,178 | Small-scale agriculture | 6.89 % (248 ha) | 41.68 % | Total (20) Class 2 (11) Class 3 (9) Class 4 (7) | 1 | Advanced | 11–14 | 25 |
| | | | | | | | | | 2 | Intermediate | 8–10 | 18 |
| | | | | | | | | | 3 | Intermediate | 8–11 | 20 |
| | | | | | | | | | 4 | Advanced | 12–14 | 22 |
| | | | | | | | | | 5 | Intermediate | 9–12 | 20 |
| | | | | | | | | | 6 | Intermediate | 10–12 | 19 |
| | | | | | | | | | 7 | Advanced | 12–15 | 25 |
| | | | | | | | | | 8 | Intermediate | 8–10 | 20 |
| Ubaíra (30 %) | 39°40′13″W 13°07′18″S | Submontane dense rainforest | 792 | 1,172 | Pastures | 26.11 % (936 ha) | 33.86 % | Total (27) Class 2 (15) Class 3 (9) Class 4 (14) | 1 | Intermediate | 10–13 | 20 |
| | | | | | | | | | 2 | Advanced | 13–16 | 40 |
| | | | | | | | | | 3 | Advanced | 13–16 | 40 |
| | | | | | | | | | 4 | Advanced | 13–16 | 40 |
| | | | | | | | | | 5 | Advanced | 12–16 | 40 |
| | | | | | | | | | 6 | Advanced | 13–16 | 40 |
| | | | | | | | | | 7 | Advanced | 11–13 | 40 |
| | | | | | | | | | 8 | Intermediate | 10–12 | 30 |
| Nilo Peçanha (35 %) | 39°11′46″W 13°38′56″S | Dense rainforest | 163 | 2,199 | Mixed: presence of pastures and small-scale agriculture | 5.38 % (193 ha) | 32.54 % | Total (33) Class 2 (18) Class 3 (11) Class 4 (7) | 1 | Intermediate | 11–13 | 22 |
| | | | | | | | | | 2 | Intermediate | 10–12 | 25 |
| | | | | | | | | | 3 | Advanced | 12–15 | 25 |
| | | | | | | | | | 4 | Advanced | 12–15 | 27 |
| | | | | | | | | | 5 | Advanced | 12–15 | 28 |
| | | | | | | | | | 6 | Advanced | 12–15 | 28 |
| | | | | | | | | | 7 | Advanced | 12–15 | 28 |
| | | | | | | | | | 8 | Advanced | 10–14 | 30 |

**Table 1** continued

| Landscape | Coordinates | Vegetation | Altitude (m) | Precipitation (mm) | Matrix (non habitat) | LPI/LPA | Landscape edge proportion | Myrtaceae richness (species) | Plot | Successional stage | Canopy height (m) | Emergent trees height (m) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Wenceslau Guimarães (40 %) | 39°42′39″W 13°33′34″S | Submontane dense rainforest | 540 | 1,149 | Small-scale agriculture | 22.6 % (810 ha) | 29.12 % | Total (57) Class 2 (39) Class 3 (24) Class 4 (12) | 1 | Advanced | 13–16 | 25 |
| | | | | | | | | | 2 | Advanced | 12–16 | 22 |
| | | | | | | | | | 3 | Advanced | 11–14 | 20 |
| | | | | | | | | | 4 | Advanced | 11–15 | 21 |
| | | | | | | | | | 5 | Intermediate | 10–14 | 20 |
| | | | | | | | | | 6 | Advanced | 12–16 | 22 |
| | | | | | | | | | 7 | Intermediate | 10–12 | 20 |
| | | | | | | | | | 8 | Intermediate | 11–12 | 20 |
| Camamu (45 %) | 39°10′01″W 14°00′06″S | Dense rainforest | 146 | 2,572 | Main small-scale agriculture. Presence of pastures | 33.3 % (1.298 ha) | 17.34 % | Total (43) Class 2 (32) Class 3 (18) Class 4 (6) | 1 | Intermediate | 9–13 | 18 |
| | | | | | | | | | 2 | Advanced | 11–14 | 23 |
| | | | | | | | | | 3 | Advanced | 12–15 | 27 |
| | | | | | | | | | 4 | Advanced | 13–16 | 30 |
| | | | | | | | | | 5 | Intermediate | 9–12 | 28 |
| | | | | | | | | | 6 | Advanced | 13–16 | 35 |
| | | | | | | | | | 7 | Intermediate | 8–10 | 23 |
| | | | | | | | | | 8 | Advanced | 13–16 | 35 |

**Table 1** continued

| Landscape | Coordinates | Vegetation | Altitude (m) | Precipitation (mm) | Matrix (non habitat) | LPI/LPA | Landscape edge proportion | Myrtaceae richness (species) | Plot | Successional stage | Canopy height (m) | Emergent trees height (m) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Jaguaripe (55 %) | 39°01′19″W 13°10′22″S | Dense rainforest | 61 | 1,937 | Main pastures. Presence of small-scale agriculture | 38.78 % (1,396 ha) | 22.02 % | Total (47) Class 2 (40) Class 3 (13) Class 4 (7) | 1 | Advanced | 12–16 | 25 |
| | | | | | | | | | 2 | Intermediate | 10–13 | 23 |
| | | | | | | | | | 3 | Advanced | 10–14 | 25 |
| | | | | | | | | | 4 | Intermediate | 8–11 | 25 |
| | | | | | | | | | 5 | Advanced | 11–14 | 27 |
| | | | | | | | | | 6 | Intermediate | 9–12 | 25 |
| | | | | | | | | | 7 | Advanced | 12–16 | 30 |
| | | | | | | | | | 8 | Intermediate | 8–11 | 21 |

*Landscape* a county located in the state of Bahia where was collected the woody plants, with the total habitat amount percentage in the landscape; *Coordinates* represents the geographical coordinates (Datum SAD 69) in the center point of the landscape; *Vegetation* type of landscape vegetation based on the classification of the Veloso et al. 1991; *Altitude* the mean altitude of the eight plots sampled in each landscape; *Precipitation* annual mean value in millimeters of the precipitation observed in the county where located the sampled landscape, based on the historical series of the Brazilian National Water Agency (ANA); *Matrix* predominant type of human use and occupation observed in non-forested landscape; *LPI/LPA* value that indicates the proportion of the landscape occupied by the largest fragment (LPI) and its size (LPA) in hectares; *Landscape edge proportion* the habitat proportion in the landscape located in edge area (the edge depth are 50 meters); *Myrtaceae richness* number of collected species in each landscape: class 1 = total richness, class 2 = species richness with CBH between 8 cm and 15 cm, class 3 = species richness with CBH between 15 and 30 cm, and class 4 = species richness with CBH ≥30 cm; *Plot* sampled plot number of each landscape; *Successional stage* vegetation quality where was allocated the plot, based on CONAMA Resolution 05/94; *Canopy height* the approximate height of the canopy sample plot, estimated with the 12 m pole; *Emergent trees height* approximate height of the tallest trees found in the sampled plot, estimated with the 12 m pole

**Table 2** List of Myrtaceae species and landscape where it was collected (habitat %)

| Scientific name | Habitat landscape (%) of occurrence | Scientific name | Habitat landscape (%) of occurrence |
| --- | --- | --- | --- |
| *Blepharocalyx* sp. 1 | 35, 45 | *Eugenia* sp. 11 | 45 |
| *Calyptranthes* cf. grandifolia | 55, 45, 40 | *Eugenia* sp. 12 | 40 |
| *Calyptranthes lucida* | 30 | *Eugenia* sp. 13 | 40 |
| *Calyptranthes* aff. lucida | 25 | *Eugenia* sp. 14 | 20 |
| *Calyptranthes* sp. nova | 45 | *Eugenia* sp. 15 | 35, 40, 45, 55 |
| *Calyptranthes* sp. 1 | 55 | *Eugenia* sp. 16 | 40, 45 |
| *Calyptranthes* sp. 2 | 45 | *Eugenia* sp. 17 | 40 |
| *Calyptranthes* sp. 3 | 40 | *Eugenia* sp. 18 | 40 |
| *Calyptranthes* sp. 4 | 30 | *Eugenia* sp. 19 | 5 |
| *Calyptranthes* sp 5 | 55 | *Eugenia* sp. 20 | 30, 55 |
| *Campomanezia* cf. reitziana | 40 | *Eugenia* sp. 21 | 40 |
| *Eugenia* cf. altissima | 5, 25 | *Eugenia* sp. 22 | 40 |
| *Eugenia* cf. astringens | 55 | *Eugenia* sp. 23 | 45 |
| *Eugenia* cf. beaurepaireana | 40 | *Eugenia* sp. 24 | 55 |
| *Eugenia* aff. blastantho | 45 | *Eugenia* sp. 25 | 40 |
| *Eugenia brasiliensis* | 40, 55 | *Eugenia* sp. 26 | 35 |
| *Eugenia* cf. candolleana | 20 | *Eugenia* sp. 28 | 30 |
| *Eugenia* cf. cymatodes | 40 | *Eugenia* sp. 29 | 55 |
| *Eugenia excelsa* | 25, 30, 40, 45, 55 | *Eugenia* sp. 30 | 40 |
| *Eugenia* aff. excelsa | 55, 20 | *Eugenia* sp. 31 | 45 |
| *Eugenia* aff. flamingensis | 25, 35, 55, 45 | *Eugenia* sp. 32 | 20 |
| *Eugenia itacarensis* | 15, 25, 55 | *Eugenia* sp. 33 | 35 |
| *Eugenia* aff. itacarensis | 55, 40 | *Eugenia* sp. 34 | 45 |
| *Eugenia* cf. itapemiririrensis | 40 | *Eugenia* sp. 35 | 35 |
| *Eugenia macrosperma* | 35, 55 | *Eugenia* sp. 36 | 40 |
| *Eugenia* cf. mandiocensis | 40, 45 | *Eugenia* sp. 37 | 25 |
| *Eugenia* aff. paracatuana | 40 | *Eugenia* sp. 38 | 25 |
| *Eugenia* aff. pauciflora | 40 | *Eugenia* sp. 39 | 55 |
| *Eugenia* cf. persicifolia | 35 | *Eugenia* sp. 40 | 35 |
| *Eugenia* aff. pleurantha | 40 | *Eugenia* sp. 41 | 40 |
| *Eugenia* aff. prasina | 20, 30, 55 | *Eugenia* sp. 42 | 30 |
| *Eugenia* cf. pruinosa | 55 | *Eugenia* sp. 43 | 40 |
| *Eugenia* cf. rostrata | 45, 55 | *Eugenia* sp. 46 | 15 |
| *Eugenia* aff. rostrata | 30, 40 | *Eugenia* sp. 47 | 20 |
| *Eugenia* sp. 1 | 20 | *Eugenia* sp. 48 | 35 |
| *Eugenia* sp. 2 | 20 | *Eugenia* sp. 49 | 45 |
| *Eugenia* sp. 4 | 45 | *Eugenia* sp. 50 | 55 |
| *Eugenia* sp. 5 | 25, 35 | *Eugenia* sp. 51 | 55 |
| *Eugenia* sp. 6 | 55 | *Eugenia* sp. 52 | 20 |
| *Eugenia* sp. 7 | 55 | *Eugenia* sp. 53 | 40 |
| *Eugenia* sp. 8 | 25, 30 | *Eugenia* sp. 54 | 55 |
| *Eugenia* sp. 9 | 35 | *Eugenia* sp. 55 | 30, 45 |
| *Eugenia* sp. 10 | 30, 35, 55 | *Eugenia* sp. Nova | 25 |

**Table 2** continued

| Scientific name | Habitat landscape (%) of occurrence | Scientific name | Habitat landscape (%) of occurrence |
| --- | --- | --- | --- |
| *Marlierea* cf. levigata | 5, 30 | *Myrcia* sp. 16 | 55 |
| *Marlierea* cf. neowiedeana | 55 | *Myrcia* sp. 17 | 30 |
| *Marlierea* obumbrans | 25 | *Myrcia* sp. nova | 30, 55 |
| *Marlierea* cf. obversa | 35, 55 | *Myrcia* sp. nova 1 | 55 |
| *Marlierea* cf. regeliana | 30, 35 | *Myrcia* sp. nova 2 | 40 |
| *Marlierea* cf. rufa | 40, 55 | *Myrciaria ferruginea* | 35, 40 |
| *Marlierea* aff. silvatica | 45 | *Myrciaria* cf. floribunda | 20, 25, 30, 40, 45, 55 |
| *Marlierea* sp. 1 | 40 | *Myrciaria tenella* | 45 |
| *Marlierea* sp. 2 | 30 | *Myrciaria* sp. 1 | 30 |
| *Marlierea* sp. 4 | 45 | *Myrciaria* sp. nova 1 | 40 |
| *Marlierea* sp. 5 | 45 | *Neomitranthes* cf. glomerata | 35, 40, 55 |
| *Marlierea* sp. 6 | 40 | *Plinia* cf. callosa | 45 |
| *Marlierea* sp. 7 | 55 | *Plinia* aff. rara | 45 |
| *Myrceugenia* cf. miersiana | 30 | *Plinia* sp. 1 | 35 |
| *Myrcia* cf. bergiana | 35, 40 | *Plinia* sp. 3 | 45 |
| *Myrcia* aff. bicolor | 40 | *Psidium* aff. myrtoides | 40 |
| *Myrcia* aff. blanchetiana | 40 | *Psidium* sp. 1 | 45 |
| *Myrcia* cf. calyptranthoides | 40 | *Psidium* sp. 2 | 20 |
| *Myrcia* colpodes | 35 | Sp. 1 | 40 |
| *Myrcia* cf. crocea | 35, 45 | Sp. 2 | 40 |
| *Myrcia* cf. hirtiflora | 5, 25 | Sp. 3 | 40 |
| *Myrcia* cf. micropetala | 45 | Sp. 4 | 40 |
| *Myrcia* cf. neoblanchetiana | 30, 40 | Sp. 5 | 40 |
| *Myrcia* pulchra | 30, 40 | Sp. 6 | 35 |
| *Myrcia* cf. racemosa | 30, 35, 40, 55 | Sp. 7 | 45 |
| *Myrcia* rethusa | 30 | Sp. 8 | 35, 45 |
| *Myrcia* aff. rethusa | 40, 55 | Sp. 9 | 35 |
| *Myrcia* silvatica | 15, 25, 45, 55 | Sp. 10 | 35 |
| *Myrcia* splendens | 5, 15, 25, 30, 35, 40, 45, 55 | Sp. 11 | 55 |
| *Myrcia* cf. springiana | 30 | Sp. 12 | 55 |
| *Myrcia* vittoriana | 25, 35, 40, 45 | Sp. 13 | 55 |
| *Myrcia* aff. vittoriana | 25, 35, 55 | Sp. 14 | 55 |
| *Myrcia* sp. 1 | 45 | Sp. 15 | 45 |
| *Myrcia* sp. 3 | 40 | Sp. 16 | 35 |
| *Myrcia* sp. 4 | 45 | Sp. 17 | 25 |
| *Myrcia* sp. 5 | 30 | Sp. 18 | 45 |
| *Myrcia* sp. 7 | 45 | Sp. 20 | 25 |
| *Myrcia* sp. 8 | 40 | Sp. 22 | 45 |
| *Myrcia* sp. 9 | 55 | Sp. 23 | 15 |
| *Myrcia* sp. 10 | 45 | Sp. 25 | 40 |

**Table 2** continued

| Scientific name | Habitat landscape (%) of occurrence | Scientific name | Habitat landscape (%) of occurrence |
|---|---|---|---|
| *Myrcia* sp. 11 | 15 | Sp. 26 | 40 |
| *Myrcia* sp. 13 | 35 | Sp. 27 | 55 |
| *Myrcia* sp. 14 | 40 | Sp. 28 | 35 |
| *Myrcia* sp. 15 | 25, 30, 35, 45, 55 | Sp. 29 | 55 |

# References

Andrén H (1994) Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos 71:355–366

Banks-Leite C, Ewers RM, Kapos V, Martensen AC, Metzger JP (2011) Comparing species and measures of landscape structure as indicators of conservation importance. J Appl Ecol 48:706–714

Bawa KS, Kress WJ, Nadkarni NM, Lele S (2004) Beyond paradise-meeting the challenges in tropical biology in the 21 st century. Biotropica 36:437–446

Bennett AF, Haslem A, Cheal DC, Clarke MF, Jones RN, Koehn JD, Lake PS, Lumsden LF, Lunt ID, Mackey BG, Nally RM, Menkhorst PW, New TR, Newell GR, O'Hara T, Quinn GP, Radford JQ, Robinson D, Watson JEM, Yen AL (2009) Ecological processes: a key element in strategies for nature conservation. Ecol Manage Restor 10:192–199

Cavalcanti C (2006) Opulência vegetal, cobiça insaciável e a entropização da entropia: uma visão da história socioambiental da Mata Atlântica. In: Siqueira-Filho JA, Leme E (eds) Fragmentos da Mata Atlântica do Nordeste—biodiversidade, conservação e suas bromélias. Andrea Jakobsson Estúdio, Rio de Janeiro, pp 13–45

CONAMA (1994) Resolução n°05 que define vegetação primária e secundária nos estágios inicial, médio e avançado de regeneração da Mata Atlântica, a fim de orientar os procedimentos de licenciamento de atividades florestais no Estado da Bahia. Diário Oficial da União 101:7912–7913

Condit R, Hubbell SP, Foster RB (1992) Recruitment near conspecific adults and the maintenance of tree and shrub diversity in a neotropical forest. Am Nat 140:261–286

de Blois S, Domon G, Bouchard A (2002) Landscape issues in plant ecology. Ecography 25:244–256

Drinnan IN (2005) The search for fragmentation thresholds in a southern Sydney suburb. Biol Conserv 124:339–349

Ewers RM, Didham RK (2006) Confounding factors in the detection of species responses to habitat fragmentation. Biol Rev 81:117–142

Fahrig L (2001) How much habitat is enough? Biol Conserv 100:65–74

Fahrig L (2002) Effect of habitat fragmentation on the extinction threshold: a synthesis. Ecol Appl 12:346–353

Fahrig L (2003) Effects of habitat fragmentation on biodiversity. Annu Rev Ecol Evol Syst 34:487–515

FAO—Food and Agriculture Organization of the United Nations (2001) State of the world's forests. FAO, Rome

Faria D, Mariano-Neto E, Martini AMZ, Ortiz JV, Montingelli R, Rosso S, Paciencia MLB, Baumgarten J (2009) Forest structure in a mosaic of rainforest sites: the effect of fragmentation and recovery after clear cut. Forest Ecol Manag 257:2226–2234

Ficetola GF, Denoël M (2009) Ecological thresholds: an assessment of methods to identify abrupt changes in species-habitat relationships. Ecography 32:1075–1084

Flather CH, Bevers M (2002) Patchy reaction-diffusion and population abundance: the relative importance of habitat amount and arrangement. Am Nat 159:40–56

Fundação SOS Mata Atlântica and INPE (2008) Atlas dos remanescentes florestais da Mata Atlântica período 2000–2005. Fundação SOS Mata Atlântica/Instituto Nacional de Pesquisas Espaciais, São Paulo

Giulietti AM, Harley RM, Queiroz LP, Wanderley MGL, van den Berg C (2005) Biodiversity and conservation of plants in Brazil. Conserv Biol 19:632–639

Gressler E, Pizo MA, Morellato PC (2006) Polinização e dispersão de sementes de Myrtaceae no Brasil. Rev Bras Bot 29:509–530

Groffman PM, Baron JS, Blett T, Gold AJ, Goodman I, Gunderson LH, Levinson BM, Palmer MA, Paerl HW, Peterson GD, Poff NL, Rejeski DW, Reynolds JF, Turner MG, Weathers KC, Wiens J (2006) Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9:1–13

Guedes-Bruni RR, Pessoa SVA, Kurtz BC (1997) Florística e estrutura do componente arbustivo-arbóreo de um trecho preservado de floresta montana na Reserva Ecológica de Macaé de Cima. In: Lima HC, Guedes-Bruni RR (eds) Serra de Macaé de Cima: diversidade florística e conservação em Mata Atlântica. Instituto de Pesquisas Jardim Botânico do Rio de Janeiro, Rio de Janeiro, pp 127–136

Hanski I (1994) A practical model of metapopulation dynamics. J Anim Ecol 63:151–162

Huggett AJ (2005) The concept and utility of ecological thresholds' in biodiversity conservation. Biol Conserv 124:301–310

King RS, Baker ME (2011) An alternative view of ecological community threshold and appropriate analyses for their detection: comment. Ecol Appl 21:2833–2839

Kuussaari M, Bommarco R, Heikkinen RK, Helm A, Krauss J, Lindborg R, Öckinger E, Pärtel M, Pino J, Roda F, Stefanescu C, Teder T, Zobel M, Steffan-Dewenter I (2009) Extinction debt: a challenge for biodiversity conservation. Trends Ecol Evol 24:564–571

Lindenmayer DB, Luck G (2005) Synthesis: thresholds in conservation and management. Biol Conserv 124:351–354

Lindenmayer DB, Fischer J, Cunningham RB (2005) Native vegetation cover thresholds associated with species responses. Biol Conserv 124:311–316

Lopes AV, Girão LC, Santos BA, Peres CA, Tabarelli M (2009) Long-term erosion of tree reproductive trait diversity in edge-dominated Atlantic forest fragments. Biol Conserv 142:1154–1165

MacArthur RH, Wilson EO (1967) The theory of island biogeography. Princeton University Press, Princeton

Martensen AC, Pimentel RG, Metzger JP (2008) Relative effects of fragment size and connectivity on bird community in the Atlantic Rain Forest: implications for conservation. Biol Conserv 141:2184–2192

Melo MMF, Mantovani W (1994) Composição florística e estrutura de trecho de Mata Atlântica de encosta na Ilha do Cardoso (Cananéia, SP, Brasil). Bol Inst Bot 9:107–158

Mendonça JR, de Carvalho AM, Mattos-Silva LA, Thomas WW (1994) 45 anos de desmatamento no sul da Bahia, remanescentes da Mata Atlântica—1945(1960), 1974, 1990. Projeto Mata Atlântica Nordeste/CEPEC, Ilhéus

Metzger JP, Martensen AC, Dixo M, Bernacci LC, Ribeiro MC, Teixeira AMG, Pardini R (2009) Time-lag in biological responses to landscape changes in a highly dynamic Atlantic Forest region. Biol Conserv 142:1166–1177

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: biodiversity synthesis. World Resources Institute, Washington DC

Montoya D, Alburquerque FS, Rueda M, Rodríguez MA (2010) Species' response patterns to habitat fragmentation: do trees support the extinction threshold hypothesis? Oikos 119:1335–1343

Mori SA, Boom BM, Carvalino AM, Santos TS (1983) Ecological importance of Myrtaceae in an Eastern Brazilian Wet Forest. Biotropica 15:68–70

Muggeo VMR (2008) Segmented: an R package to fit regression models with broken-line relationships. R News 8:20–25

Muggeo VMR (2012) The package segmented 0.2–8.4. http://www.R-project.org. Accessed 20 Apr 2012

Murray-Smith C, Brummitt NA, Oliveira-Filho AT, Bachman S, Moat J, Lughadha EMN, Lucas EJ (2008) Plant diversity hotspots in the Atlantic coastal forests of Brazil. Conserv Biol 23:151–163

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Oliveira LC, Hankerson SJ, Dietz JM, Raboy BE (2010) Key tree species for the golden-headed lion tamarin and implications for shade-cocoa management in southern Bahia, Brazil. Anim Conserv 13:60–70

Paltto H, Norden B, Gotmark F, Franc N (2006) At which spatial and temporal scales does landscape context aff.ect local density of Red Data Book and Indicator species? Biol Conserv 133:442–454

Pardini R, de Souza SM, Braga-Neto R, Metzger JP (2005) The role of forest structure, fragment size and corridors in maintaining small mammal abundance and diversity in an Atlantic forest landscape. Biol Conserv 124:253–266

Pardini R, Faria D, Accacio GM, Laps RR, Mariano-Neto E, Paciencia MLB, Dixo M, Baumgarten J (2009) The challenge of maintaining Atlantic forest biodiversity: a multi-taxa conservation assessment of specialist and generalist species in an agro-forestry mosaic in southern Bahia. Biol Conserv 142:1178–1190

Pardini R, Bueno AA, Gardner TA, Prado PI, Metzger JP (2010) Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. PLoS One 5:1–10

Primack RB, Rodrigues E (2001) Biologia da conservação. Editora Planta, Paraná

R Development Core Team (2012) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna

Radford JQ, Bennett AF, Cheers GJ (2005) Landscape-level thresholds of habitat cover for woodland-dependent birds. Biol Conserv 124:317–337

Rhodes JR, Callagham JG, McAlpine CA, de Jong C, Bowen ME, Mitchell DL, Lunney D, Possingham HP (2008) Regional variation in habitat–occupancy thresholds: a warning for conservation planning. J Appl Ecol 45:549–557

Ribeiro MC, Metzger JP, Martensen AC, Ponzoni FJ, Hirota MM (2009) The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. Biol Conserv 142:1141–1153

Rigueira DMG (2011) Limiares ecológicos na economia pós-moderna. Ciência Hoje 48:68–69

Rigueira DMG, Coutinho SL, Pinto-Leite CM, Sarno VLC, Estavillo C, Chastinet CBA, Dias VS, Campos S (2013) Perda de habitat, leis ambientais e conhecimento científico: proposta de critérios para a avaliação de pedidos de supressão de vegetação. Rev Caititu 1:01–22. doi:10.7724/caititu.2012.v1.n1.d03

Rodrigues RR, Lima RAF, Gandolfi S, Nave AG (2009) On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. Biol Conserv 142:1242–1251

Santos BA, Peres CA, Oliveira MA, Grillo A, Alves-Costa CP, Tabarelli M (2008) Drastic erosion in functional attributes of tree assemblages in Atlantic Forest fragments of northeastern Brazil. Biol Conserv 141:249–260

Schmidt BD, Roland J (2006) Moth diversity in a fragmented habitat: importance of functional groups and landscape scale in the boreal forest. Ann Entomol Soc Am 99:1110–1120

Silva AF, Leitão-Filho HF (1982) Composição florística e estrutura de um trecho de mata de encosta no município de Ubatuba. Rev Bras Bot 5:43–52

Silva GC, Nascimento MT (2001) Fitossociologia de um remanescente de mata sobre tabuleiros no norte do estado do Rio de Janeiro (Mata do Carvão). Rev Bras Bot 24:51–62

Souza VC, Lorenzi H (2005) Botânica sistemática: guia ilustrado para identificação das famílias de Angiospermas da flora brasileira, baseado em APG II. Instituto Plantarum, São Paulo

Stehmann JR, Forzza RC, Salino A, Sobral M, Costa DP, Kamino LHY (2009) Plantas da floresta atlântica. Instituto de Pesquisas Jardim Botânico do Rio de Janeiro, Rio de Janeiro 516p

Swift TL, Hannon SJ (2010) Critical thresholds associated with habitat loss: a review of the concepts, evidence, and applications. Biol Rev 85:35–53

Tabarelli M, Peres CA (2002) Abiotic and vertebrate seed dispersal in the Brazilian Atlantic forest: implications for forest regeneration. Biol Conserv 106:165–176

Tabarelli M, da Silva JMC, Gascon C (2004) Forest fragmentation, synergisms and the impoverishment of neotropical forests. Biodivers Conserv 13:1419–1425

Tabarelli M, Aguiar AV, Ribeiro MC, Metzger JP (2010) Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. Biol Conserv 143:2328–2340

Tilman D, May RM, Lehman CL, Nowak MA (1996) Habitat destruction and the extinction debt. Nature 371:65–66

Toms JD, Lesperance ML (2003) Piecewise regression: a tool for identifying ecological thresholds. Ecology 84:2034–2041

Turner IM (1996) Species loss in fragments of tropical rain forest: a review of the evidence. J Appl Ecol 33:200–209

United Nations (1992) Agenda 21: The UN programme of action from Rio. United Nations, New York

Veloso HP, Rangel-Filho ALR, Lima JCA (1991) Classificação da vegetação brasileira, adaptada a um sistema universal. IBGE/Departamento de Recursos Naturais e Estudos Ambientais, Rio de Janeiro

Wright SJ, Muller-Landau HC (2006) The future of tropical forest species. Biotropica 38:287–301