

What is the value of eucalyptus monocultures for the biodiversity of the Atlantic forest? A multitaxa study in southern Bahia, Brazil

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Abstract: Eucalyptus plantations are increasing in Brazil, frequently replacing pastures, but there is still scarce information about its capacity to maintain the fauna of neighbor forest remnants. In this study, we compared descriptors of the communities of leaf litter organisms (lizards, anurans, myriapods, arachnids, orthopterans, coleopterans, and ants) between a large remnant of primary Atlantic Forest and an adjacent eucalyptus monoculture (phase 1). Then, we compared the same descriptors for leaf litter lizards and anurans, Euglossini bees, and frugivorous butterflies among the largest remnant, small remnants at intermediate regeneration stage, and eucalyptus monocultures that were not adjacent to the largest remnant (phase 2). Monocultures were sampled immediately before logging. In phase 1, we detected significant differences in structure between the forest and the monoculture in six out of seven communities sampled. Ca. 81% of the species of the landscape were recorded in the forest, but only 54% of these were found also in the monoculture. In phase 2, the structure of two out of four forest communities was significantly different from the structure of small remnants and monocultures. On average, 76% of the species found in the whole landscape were sampled in the forest. Out of this subset, on average 74% of the species were also sampled in small remnants and 68% in monocultures. Findings of the present study point out a moderate capacity of eucalyptus monocultures to harbor species of the forest fauna even when fully grown but highlights the opportunity that they might offer for increasing connectivity in anthropogenic forest landscapes depending on their management.

Keywords: Atlantic forest; monoculture; biodiversity

Introduction

The destruction and fragmentation of tropical forests are among the anthropogenic factors that contribute most for biodiversity erosion in the planet (Prugh et al. 2008). Part of this erosion originates from population processes that act after forest fragmentation (Saunders et al. 1991). Local extinctions, for example, may not be followed by recolonization if the environments located between fragments act as barriers to the dispersal of forest organisms. It is expected, hence, that in most cases when these environments are suitable for forest organisms, extinctions will be buffered (Vandermeer and Carvajal 2001). Therefore, a great concern of biodiversity management in landscapes with forest remnants is the quality assessment of anthropogenic environments in terms of their capacity to allow survival and dispersal of forest species.

Silviculture (e.g., palm, pine, eucalyptus, and cacao plantations) has been undergoing a great expansion in the whole world, from over 130 million ha in 1996 to 187 million ha in 2001 (FAO 2001). Silviculture produces anthropogenic environments that, due to their forest-like physiognomy, could at least temporarily increase functional connectivity between forest remnants and minimize deleterious effects of fragmentation, in comparison to non-forest matrices such as pastures and other plantations (Lindenmayer and Hobbs 2004). Eucalyptus monocultures represented, in the beginning of this century, 10% of the total silviculture area of the world, including over 20 million ha distributed over almost 100 countries (FAO 2001). In Brazil, eucalyptus monocultures represented 62% of the agroforests, summing up 3.55 million ha. More than one third of these areas are associated with the cellulose and paper industry, which had a revenue of US\$ 13.7 billion in 2006 and represented the highest share of forest products in Brazilian exports in that year (US\$ 4 billion, ca.

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3% of all exports of the country in 2006), turning the country into the largest exporter of eucalyptus cellulose fiber in the world (SBS 2007). The largest sustained production of eucalyptus timber in Brazil is located in the southeastern, northeastern and southern regions (SBS 2007), frequently in landscapes with Atlantic Forest remnants.

In southern Bahia State, northeastern Brazil, one of the most biodiverse areas of the world (Thomas et al. 1998), eucalyptus monocultures expanded greatly in the 1990s and 2000s, replacing pastures created in the 1980s, which were located in areas that had been logged during the three previous decades (Coimbra-Filho et al. 1996; Dean 1998). The expansion of eucalyptus monocultures in this region is determined, among other factors, by its high productivity (one of the highest in the world), which is related to climate and soil conditions. Currently, the state of Bahia has the third largest planted area of eucalyptus in Brazil (over 550 thousand ha, 15% of the planted area in the country), and these figures are expected to be increased (ABRAF 2008).

Until recently, though, there were no formal investigations that assessed the capacity of eucalyptus monocultures to shelter species of the tropical forest fauna. If this capacity is high, eucalyptus monocultures could contribute to functional connectivity in fragmented landscapes, and, hence, contribute to the persistence of populations of regional species. However, if the percentage of forest fauna in eucalyptus monocultures is low, these agroforests, as they are currently managed, should not be considered as an adequate land-use alternative that ensures functional connectivity between forest remnants in the region. Two recent papers (Barlow et al. 2007; Fonseca et al. 2004) compared the descriptors of several ecological communities between natural forests (in Amazon and Araucaria Forests, respectively) and eucalyptus monoculture. They concluded that eucalyptus monoculture cannot be considered tree deserts, but that they harbor considerably impoverished biodiversity when compared to the reference forests.

The present study aimed at assessing the capacity of eucalyptus monocultures to harbor forest species, based on a faunistic analysis of a landscape in southern Bahia, where these agroforests are associated with Atlantic Forest remnants. In the first phase, we compared the fauna of one of the largest forest remnants in the region with the fauna of an adjacent eucalyptus monoculture. In the second phase, we compared the fauna of this large fragment and of smaller fragments with the fauna of eucalyptus monocultures close to the smaller fragments.

Material and methods

The present study was carried out in the properties of the company Veracel Celulose S.A., in southern Bahia, within the municipalities of Eunápolis, Porto Seguro, and Santa Cruz Cabrália (16°20'S–39°15'W). The local climate (type Af according to Köppen's classification) is hot and humid, characteristically coastal, without a dry season, with high and quite constant temperatures, and with the highest rainfall between July and October. The topography is characterized by wide plateaus intersected by valleys associated with a dense hydrological system that in sev-

eral parts harbors regenerating forests (Fig. 1A). During the study, the eucalyptus plantations covered ca. 70,000 ha, and were almost exclusively associated with plateaus (Fig. 1B). According to the company's database, most Atlantic Forest remnants in the region had between 50 and 250 ha and were at an intermediate regeneration stage (Fig. 1C). There is still in the region one of the largest and best preserved Atlantic Forest remnants of Bahia, the preserve Estação Veracel, with 6,069 ha of primary forest and some second-growth areas at an advanced regeneration stage (Fig. 1D).

The first phase of the study aimed at comparing the fauna of the preserve Estação Veracel, the main forest remnant in the area and considered as the best local reference of the original habitat, with the fauna of a contiguous area used for eucalyptus monoculture (Fig. 1E). In this situation, if the eucalyptus monoculture is a suitable habitat for the forest fauna, it would be benefited by the proximity to a large source area, and so was expected to harbor much of the fauna observed in the forest. The second phase aimed at comparing the fauna of smaller forest remnants (50 to 250 ha, at intermediate regeneration stage) with the fauna of eucalyptus monocultures that are distant from the main source area, again using Estação Veracel as a reference. With this sampling design, it was possible to compare small forest remnants and eucalyptus monocultures in terms of their capacity to harbor elements of the local forest fauna.

In both phases, we could not delineate the experiment in order to have real replicates of the reference system, as Estação Veracel is the only large remnant in the landscape. Similarly, in the first phase we could not get real replicates of eucalyptus monoculture close to the reference system, as, at that moment, there was only one block of monoculture in such condition. However, we believe that our results on the descriptors of communities should reflect more the effects of the differences among the physiognomies of the sampled areas than the position effect, as argued below.

In both phases, we chose the oldest eucalyptus monocultures available, which were planted seven years before. At this stage of growth, trees reach their highest height and are immediately logged. An understory with native species is well developed by then due to the management system, that do not remove the native species that grow among the eucalyptus trees. The eucalyptus are planted every 3 m along rows spaced by 3 m. We restricted sampling to the fauna of plateaus, since eucalyptus monocultures are not placed in valleys.

The study groups included arthropods and vertebrates. In the first phase of the study, we sampled communities of leaf litter lizards, anurans and arthropods (Costa 2004). We carried out sampling in October 2001 and March 2002 in eight plots: four in Estação Veracel and four in the eucalyptus monoculture (Fig. 2). In order to graphically evaluate whether the attributes of the communities were responding to distance between plots rather than to the physiognomy (due to lack of real replicates in the landscape), we located half of the plots per treatment quite close to each other (E1, E3, M3, and M4 in Fig. 2). Each plot consisted of a grid with 36 pitfalls, with transects 20.4 m away from each other, each transect with 12 traps 10.2 m away from each

other. Each trap consisted of a plastic bucket (40 cm in height and 30 cm in diameter) with two 1.5-m drift fences. We checked the traps in all plots daily (10 days in each of the two sampling sessions), capturing all anurans and lizards found in all traps, as well as the arthropods found in nine traps picked at random in

the plot for three days in each sampling session. A plot that captured vertebrates in a given day was not used to the capture of arthropods in that day in order to avoid the interference in the sampling due to predation inside the traps.

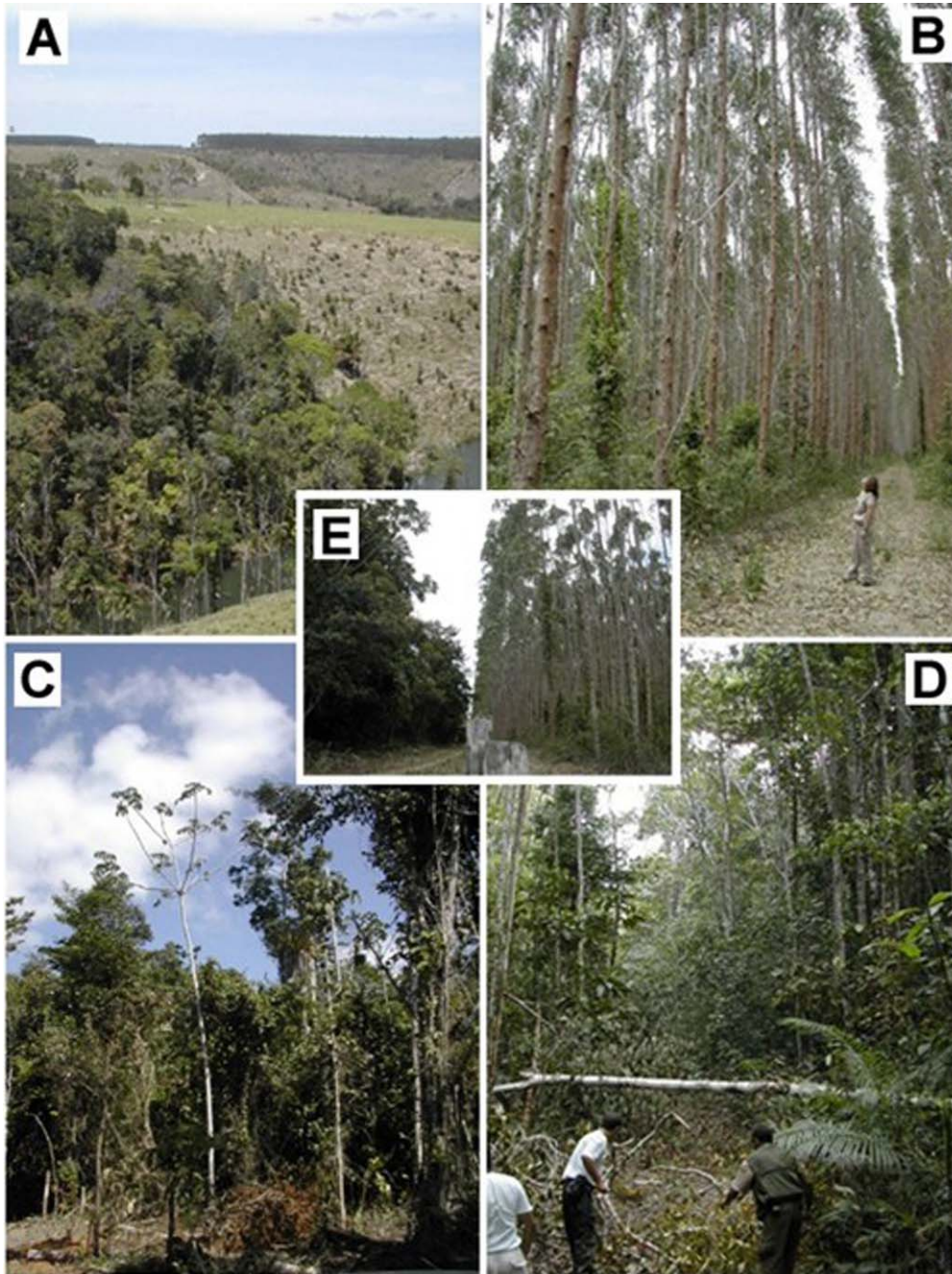


Fig. 1 Photos of our study areas in southern Bahia, Brazil. (A) General view, showing plateaus with eucalyptus plantations interspersed with valleys, some of which contain regenerating forests; (B) an approximately seven-year old eucalyptus monoculture with understory growth; (C) the edge of a forest remnant of the most common type in the region: 50 to 200 ha of extension and at an intermediate regeneration stage; (D) a trail within the preserve Estação Veracel, the largest Atlantic Forest remnant in the region, which has 6,069 ha of primary forest with portions at an advanced regeneration stage; (E) the ecotone between Estação Veracel and a eucalyptus plantation sampled in phase I of the present study.

In the second phase of the study, we sampled communities of leaf litter anurans and lizards (Dantas 2004), Euglossini bees

(Melo 2005), and frugivorous butterflies (Vasconcelos 2008). We carried out faunistic sampling in February-March 2003,

June–July 2003, September–October 2003, and December 2003–January 2004 in 12 plots: four in Estação Veracel, four in forest remnants and four in eucalyptus monocultures (Fig. 2). Sampling of vertebrates was based on grids identical to the grids used in the first phase, with daily checking of traps in all plots for ten consecutive days per sampling session, capturing all anurans and lizards. Sampling of bees was based on five collecting days per sampling session, using per plot 18 aromatic traps with essences of eucalyptol, vanilla and methyl salicylate (Neves and Viana 1997). Sampling of frugivorous butterflies was made using five traps per plot (total of 60 traps) of the type Van Someren Rydon (DeVries 1987), baited with fermented fruits, and placed 25 m away from each other.

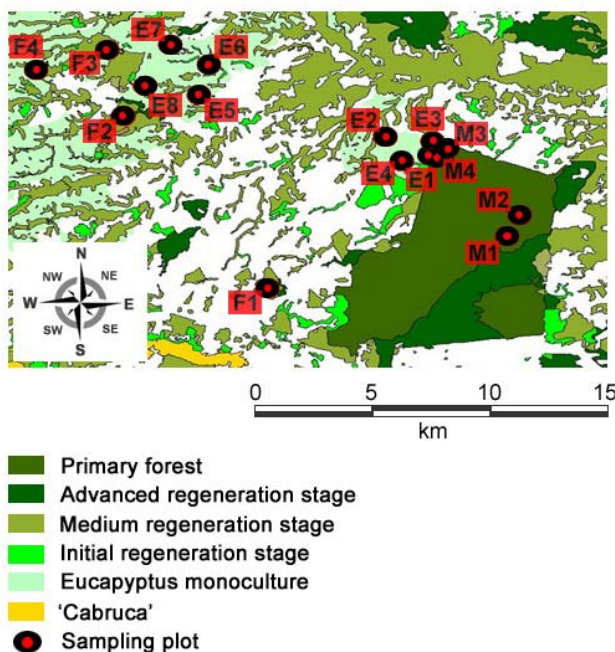


Fig. 2 The anthropogenic Atlantic Forest landscape where the present study was carried out (16°20'S - 39°15'W, in the center). At the bottom right corner of the image is the main forest remnant of the preserve Estação Veracel, showing the four sampling plots (M1 to M4). To the left of the preserve are the four plots in smaller forest fragments (F1 to F4) and the plots in adjacent eucalyptus monocultures (E1 to E4) located far from the preserve (E5 to E8). Image produced from the database of Veracel Celulose S.A.

All vouchers were deposited in the zoological collection of the Department of Zoology, Universidade Federal da Bahia. Anurans, lizards, bees, and butterflies were identified to species by specialists. We identified leaf litter arthropods (myriapods, arachnids excluding spiders, orthopterans, coleopterans, and ants) to morphospecies. We built matrices of total abundance (summing up all captures in all days of all sampling sessions) of species or morphospecies per plot. For ants, total abundance was replaced by occurrence in traps (values between 0 and 36 per plot).

We tested the hypothesis that the structure of each community differs among habitats, for each phase separately. In order to do

so, we used the permutation test MRPP (multiple response permutation procedure), calculating the effect size with the statistic A . When $A = 0$, the structure of communities is equal in all habitats. When $A = 1$, the structure of communities differs maximally among habitats (McCune and Grace 2002). As the sample size was relatively small (four plots per habitat), statistic significance tends to mean biological significance. The tests were performed using relative abundance matrices (value of each cell divided by row total) so that the relative abundance of each species per plot, and not its total abundance per plot, was assessed in the test. We used as a measure of dissimilarity between plots the Sorensen distance. To represent in two dimensions the differences in community structure among plots, we used non-metric multidimensional scaling (NMDS). In this procedure we used the matrices of relative abundance and Sorensen distances. We completed the MRPP test with an indicator species analysis (McCune and Grace 2002), which detects, through a randomization test, the species in the community that are indicators of habitats.

In MPRR test we used a significance level of 5%, but we accepted slightly higher values (i.e., $p < 0.075$) as marginally significant. We assumed in this study that the Type II error would be worse, as it could lead to the false conclusion that the eucalyptus monoculture in southern Bahia has a good capacity of harboring forest species. Since the eucalyptus monocultures sampled were assumed to be in better condition to harbor forest species (immediately before logging), during the seven years of growth their conditions must have been more unfavorable. We used a significance level of 5% ($p < 0.075$ as marginally significant) to make a decision in each test related to the indicator species analysis. As several tests were performed for each ecological community, the family-wise Type I error for each community is increased and the family-wise Type II error is decreased (Quinn and Keough 2002). As our intent with these analyses is to detect species that could present high fidelity to each habitat, we prefer to have the Type II error rate diminished. Therefore, we decided not to use any procedure to adjust significance levels to control for Type I error inflation.

We compared richness and total abundances of communities across habitats using the Mann-Whitney U test (phase 1) and the Kuskal-Wallis H test (phase 2). We used a significance level of 5% ($p < 0.075$ as marginally significant) to make a decision in each test. We did not adjust significance levels due to the motives presented above.

Results

First phase

Lizards: We collected 97 lizards of 12 species. Both mean abundance and mean richness were higher in the main forest remnant (M) than in the eucalyptus monoculture (E) (Table 1). We recorded nine species in M, out of which four were also recorded in E (Fig. 3). The MRPP test detected a significant difference in community structure between M and E ($A = 0.227$, $p = 0.011$) (Fig. 4), and two species associated with M were considered as

indicators (*E. catenatus pictus*, $p = 0.027$; *L. scincoides*, $p = 0.058$). The other species had p values higher than 0.423. Only

one or two individuals were recorded for 8 species (4 in M, 3 in E and 1 shared by M and E).

Table 1. Results of phase 1 of our study. In each cell, number of individuals (above) and of species (below, in parenthesis) of seven ecological communities sampled in four plots in the preserve Estação Veracel, the main Atlantic Forest remnant in the landscape (M1 to M4), and in four plots in a eucalyptus monoculture adjacent to the preserve (E1 to E4). Totals per habitat and the grand total are presented. P value represents the probability resulting from the Mann-Whitney U test comparing mean values per habitat (* indicates values considered significant).

Communities	Plots in the large Atlantic Forest remnant					Plots in the adjacent eucalyptus monoculture					Grand total	P value
	M1	M2	M3	M4	Total	E1	E2	E3	E4	Total		
Lizards	16 (5)	13 (4)	16 (5)	19 (7)	64 (9)	13 (5)	10 (2)	5 (3)	5 (2)	33 (7)	97 (12)	.027 * (.074)*
Anurans	8 (3)	13 (4)	10 (4)	11 (3)	42 (7)	12 (4)	83 (7)	19 (6)	33 (7)	147 (9)	189 (12)	.043 * (.036)*
Myriapods	2 (2)	12 (6)	2 (2)	9 (4)	25 (10)	8 (4)	16 (5)	15 (2)	47 (5)	86 (6)	111 (10)	.081 (.653)
Arachnids (spiders excluded)	17 (8)	5 (5)	6 (2)	0 (0)	28 (8)	2 (2)	4 (2)	3 (2)	6 (6)	15 (6)	43 (8)	.468 (.878)
Orthopterans	87 (9)	67 (7)	67 (7)	81 (6)	302 (11)	29 (6)	14 (3)	17 (5)	14 (8)	74 (11)	376 (14)	.019 * (.189)
Coleopterans	9 (7)	8 (8)	10 (9)	15 (12)	42 (26)	6 (5)	4 (3)	6 (6)	14 (8)	30 (15)	72 (35)	.146 (.059)*
Ants	27 (8)	48 (15)	15 (10)	39 (12)	129 (22)	31 (14)	26 (11)	43 (14)	42 (12)	142 (21)	271 (27)	.020 * (.465)

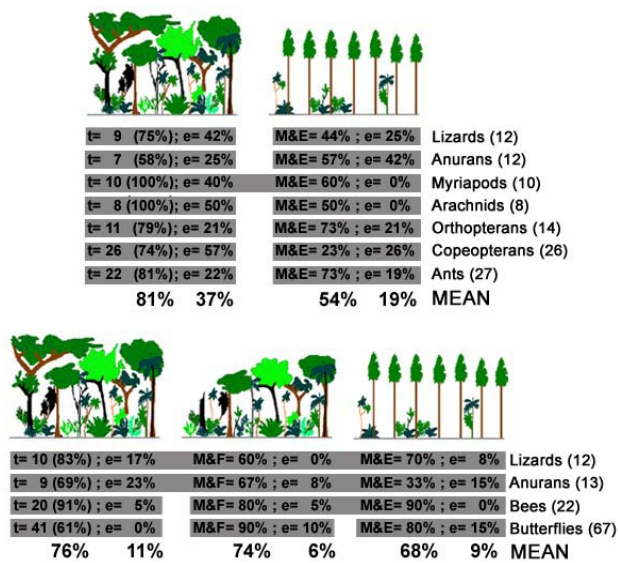


Fig. 3 Species distribution of vertebrates and arthropods in habitats in the two study phases of the present study. Above, first phase, in which we compared the fauna of the main remnant (Estação Veracel, considered as a reference forest – left) with the fauna of an adjacent eucalyptus monoculture (right). Below, second phase, in which we compared the fauna of the main remnant (left) with the faunas of smaller fragments (center) and a eucalyptus monoculture distant from the main reference forest (right). Legend: t = Total number of species recorded in the main remnant (percentage of the landscape fauna in parenthesis); e = percentage of species recorded exclusively in each habitat; $M\&F$ = percentage of species recorded in the main remnant that were also recorded in the fragments; $M\&E$ = percentage of species recorded in the main remnant that were also recorded in the eucalyptus monoculture; **number in parenthesis next to the community** = total number of species recorded in the landscape; **average** = average of percentage distributions in each phase; **gray bands** = indicate whether there is a significant difference in community structure among habitats (dashed) or not (continuous).

Anurans: We collected 189 anurans of 12 species. Mean abundance in E was higher than in M, and in one E plot the abundance was much higher than in others due to several captures of two species (*Stereocyclops incrassatus* and *Physalemus gr. cuvieri*). Mean richness was also higher in E (Table 1). We recorded seven species in M, four of them were also recorded in E (FIGURE 3). The MRPP test detected a marginally significant difference between the habitats ($A = 0.077$, $p = 0.057$) (Fig. 4). Two species associated with E were considered as indicators: *B. crucifer* ($p = 0.030$) and *Leptodactylus mystacinus* ($p = 0.030$). The other species generated p values higher than 0.143. Only one or two individuals were recorded for three species (2 in M and 1 in E).

Myriapods: We collected 111 myriapods of 10 morphospecies (8 millipedes and 2 centipedes). Mean abundance and richness did not differ between habitats, though total richness was higher in M (Table 1). All the 10 morphospecies sampled were recorded in M, and 4 of them were also recorded in E (Fig. 3). The MRPP test did not detect significant differences between habitats ($A = 0.059$, $p = 0.137$) (Fig. 4), and only one centipede morphospecies associated with E was considered as an indicator ($p = 0.055$). The other morphospecies had p values higher than 0.134. Four morphospecies were represented by only one or two individuals, all recorded in M.

Arachnids (excluding spiders): We collected 43 specimens of 8 morphospecies (4 mites, 3 harvestmen, and 1 scorpion). Mean richness and abundance were not different between habitats, through the total richness in M was twice the richness in E (Table 1). The eight morphospecies sampled were recorded in M, and 4 in E (Fig. 3). The MRPP test detected a significant difference between habitats, though with a low value of the statistic ($A = 0.092$, $p = 0.018$) (Fig. 4). No morphospecies were considered as indicators (p values higher than 0.153). Only one morphospe-

cies, present in M, was represented by one or two individuals.

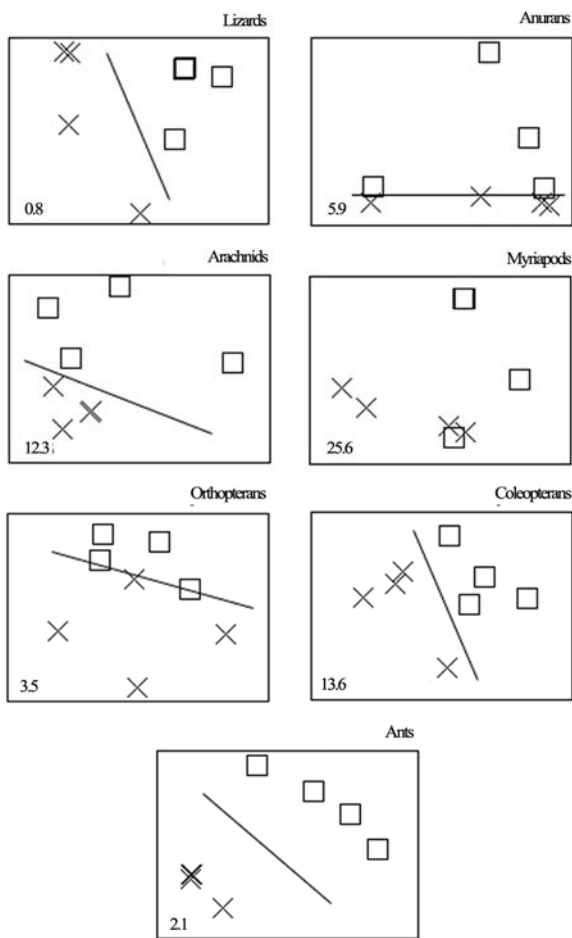


Fig. 4 Differences in community structure of leaf litter vertebrates and arthropods sampled in four plots in a large Atlantic Forest remnant (Estação Veracel - square) and in four plots of a eucalyptus monoculture adjacent to the preserve (x). The graphs represent an indirect ordination of each community by NMDS (the stress associated with the process of dimension reduction is indicated on the bottom left corner of each graph). In communities in which the MRPP test detected significant differences between habitats, we added a line separating the plots.

Orthopterans: We collected 376 specimens of 14 morphospecies (8 true crickets, 3 batids, 1 acridids, 1 phasmids, and 1 mantids). Abundance in each plot was always much higher in M than in E, though mean richness was not different between habitats (Table 1). We recorded 11 morphospecies in M, out of which 8 were present in E (Fig. 3). The MRPP test detected a significant difference between M and E ($A = 0.164$; $p = 0.025$), but no morphospecies were considered as indicators (p values higher than 0.170) (Fig. 4). Only 3 morphospecies were represented by one or two individuals, one in M and two in E.

Coleopterans: we collected 72 specimens of 35 morphospecies. Mean abundance did not differ between habitats but mean richness was higher in M than in E, and total richness in M was

much higher than in E (Table 1). We recorded 26 morphospecies in M, but only 6 of them were also detected in E (Fig. 3). The MRPP test detected a significant difference between M and E, though with a low value of the statistic ($A = 0.082$; $p = 0.017$) (Fig. 4). No morphospecies were considered as indicators (p values higher than 0.144). Twenty morphospecies were found only in M, and 9 were found only in E.

Ants: We obtained 271 records of ants of 27 morphospecies. Mean abundance was higher in M but mean richness did not differ between habitats (Table 1). We recorded 22 morphospecies in M, out of which 16 also occurred in E (Fig. 3). The MRPP test detected a significant difference between M and E ($A = 0.240$, $p = 0.006$) (Fig. 4) and 4 morphospecies were considered as indicators (p values lower than 0.057), two in M and two in E. In total, 6 morphospecies were found only in M and 5 only in E.

The position of the plots in the ordination graphs (Fig. 4) suggests that the geographic distance among them is not the main factor influencing the attributes of the studied communities, otherwise we would expect to find four plots (two from each level of the factor) closer to each other than to the plots of the respective habitat. Therefore, we interpret the differences between groups as a result of the differences in habitat properties.

In summary, six of the seven communities sampled in phase 1 exhibited different composition between M and E, and in half of them effect size was high. Out of all species sampled in the landscape, between 50% and 100% (average = 81%) were present in M, and, on average, half of this subset (from 23% to 73%) were also recorded in E. The species sampled exclusively in M varied from 21% to 57% of the total recorded for the landscape (average = 37%), and the species sampled exclusively in E varied from 0 to 42% (average = 19%) (Fig. 3).

Second phase

Lizards: We collected 206 lizards of 12 species. Mean abundance per plot was higher in the main forest remnant (M) than in the smaller fragments (F) and in the eucalyptus monoculture (E), but mean richness did not differ among habitats (Table 2). We recorded 10 species in M, out of which 6 were found in F and 7 in E (Fig. 3). The MRPP test did not detect significant differences in community structure among habitats ($A = 0.013$, $p = 0.348$) (Fig. 5), and one species associated to M was considered as an indicator (*E. catenatus pictus*, $p = 0.055$). The other species had p values higher than 0.135. Only one or two individuals were recorded for three species, one exclusively in M and one exclusively in E.

Anurans: We collected 523 anurans of 13 species. Mean abundance and richness were much higher in M and F than in E. A single species (*Chiasmocleis schubarti*) had high abundance in a single plot of M and of F. We recorded 13 species in M, out of which 6 were also found in M and 3 in E (Fig. 3). The MRPP test did not detect significant differences among habitats ($A = -0.012$, $p = 0.482$) and no species were considered as indicators (p values higher than 0.180) (Fig. 5). Five species were represented by one or two individuals: one exclusively in M, one in F and 2 in E.

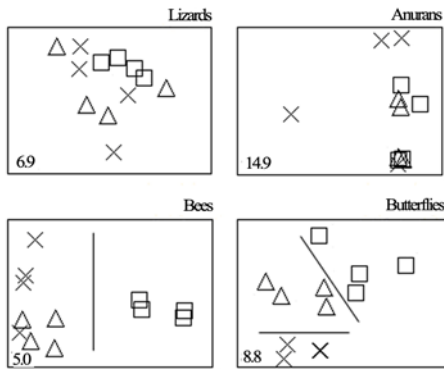


Fig. 5 Differences in community structure of vertebrates and arthropods sampled in four plots in a large Atlantic Forest remnant (Estação Veracel - square), in four plots of smaller forest fragments (triangles) and in four plots of a eucalyptus monoculture away from the RPPN (x). The graphs represent an indirect ordination of each community by NMDS (the stress associated with the process of dimension reduction is indicated on the bottom left corner of each graph). In communities in which the MRPP test detected significant differences between habitats, we added a line separating the plots.

Bees: We collected 3,872 individuals of 22 species. Mean abundance per plot was very high in M, intermediate in F and low in E. Mean richness was higher in M than in F and E, but

total richness was similar in all habitats (Table 2). We recorded 20 species in M, out of which 16 also occurred in F and 18 in E (Fig. 3). The MRPP test detected a significant difference among M, F and E ($A = 0.402, p = 0.001$), which reflects the difference found between M and F ($A = 0.432, p = 0.006$) and M and E ($A = 0.434, p = 0.006$). The difference between F and E was not significant ($A = 0.068, p = 0.071$) (Fig. 5). The indicator species analysis detected 13 species characteristic of M (p lower than 0.060). Only one species (in F) was recorded based on one or two individuals.

Butterflies: We collected 6,171 individuals of 67 species. Mean abundance per plot was very high in E, intermediate in E and low in M, and mean richness was higher in R and F than in M (Table 2). We recorded 41 species in M, 37 of which were detected in F and 33 in E (Fig. 3). The MRPP detected a significant difference among habitats ($A = 0.339, p < 0.001$), which reflects differences among all pairs of units, though effect size is small in the comparison MxF (MxF: $A = 0.339, p < 0.001$; MxE: $A = 0.395, p = 0.006$; FxE: $A = 0.351, p = 0.006$) (FIGURE 5). The indicator species analysis detected 7 species characteristic of F and 13 of E (p lower than 0.066). No butterfly species was indicator of M. Only one or two individuals were recorded for 18 species, three exclusively in M, nine in F and three in E.

Table 2. Results of phase 2 of our study. in each cell, number of individuals (above) and of species (below, in parenthesis) of four ecological communities sampled in four plots in the preserve Estação Veracel, a large Atlantic Forest remnant (M1 to M4), in four plots located in smaller forest fragments (F1 to F4) and in four plots of eucalyptus monocultures distant from the preserve (E5 to E8). Totals per habitat and the grand total are presented. p value represents the probability resulting from the Kruskal-Wallis H test comparing mean values per habitat (* indicates values considered significant). The letters a, b and c (in total columns) represent homogeneous groups of number of individuals or number of species for a given community in different habitats.

Communities	Plots in the large Atlantic Forest remnant					Plots in the small Atlantic Forest remnants					Plots in eucalyptus monocultures distant from the large remnant					Grand Total	P value
	M1	M2	M3	M4	Total	F1	F2	F3	F4	Total	E5	E6	E7	E8	Total		
Lizards	23 (6)	20 (6)	29 (4)	26 (5)	98 a (10) a	20 (4)	15 (6)	16 (3)	8 (2)	59 b (7) a	5 (3)	17 (5)	11 (5)	16 (5)	49 b (9) a	206 (12)	.029 * (.313)
Anurans	5 (2)	223 (7)	10 (3)	18 (5)	256 a (9) a	203 (6)	14 (2)	30 (3)	14 (4)	261 a (8) a	1 (1)	2 (2)	2 (2)	1 (1)	6 b (5) b	523 (13)	.032 * (.051) *
Bees	817 (18)	1096 (20)	472 (19)	325 (18)	2710 a (20) a	213 (16)	146 (14)	231 (13)	129 (10)	719 b (18) b	68 (12)	140 (14)	110 (11)	125 (12)	443 c (19) b	3872 (22)	.010 * (.021) *
Butterflies	53 (12)	32 (15)	112 (27)	108 (29)	305 a (41) a	641 (45)	326 (35)	185 (42)	199 (41)	1351 b (58) b	1068 (35)	1125 (37)	792 (34)	1530 (36)	4515 c (49) b	6171 (67)	.007 * (.014) *

In summary, two out of four communities sampled in phase 2 exhibited significantly different composition between M and the other habitats, and in both effect size was large. Among all species sampled in the landscape, between 61 and 91% (average 76%) were present in M, and on average three quarters of this subset were present in F (from 60% to 90%) and two thirds in E (from 33% to 90%). The species sampled exclusively in M varied from 0 to 23% of the total recorded for the landscape (average = 11%), in F it varied from 0 to 10% (average = 6%) and in E from 0 to 15% (average = 9%) (Fig. 3).

Discussion

In the first phase of our study, we tried to assess whether a eucalyptus monoculture located very close to a large source area of Atlantic Forest was able to harbor a considerable portion of the forest fauna. The comparison carried out between these two habitats detected significant differences in the structure of six out of seven communities sampled. The forest exhibited, on average, 37% of exclusive species, and the eucalyptus monoculture maintained on average half of the species present in the forest. In the

second phase, we tried to assess whether a eucalyptus monoculture away from a large source area was able to harbor a considerable portion of the forest fauna, comparing its performance with that of small forest remnants in the region at an intermediate regeneration stage. The comparison carried out among habitats detected significant differences in the structure of two out of four communities sampled; the forest exhibited, on average, 11% of exclusive species, the fragments maintained ca. three quarters of the fauna present in the forest, and the eucalyptus monocultures ca. two thirds of the forest fauna.

Our results are very similar to those of Barlow and collaborators, who compared 15 animal and plant communities in eucalyptus plantations, forests in regeneration, and primary forests in the Amazon (Barlow et al. 2007). However, at first sight they seem to differ somewhat from the results from Fonseca and collaborators, who compared 13 fungi, plant, and animal communities in *Pinus*, *Araucaria* and *Eucalyptus* plantations and Araucaria Forest (a subtropical rain forest located on the south most part of the Atlantic Forest) in southern Brazil (Fonseca et al. 2009). The average number of species recorded exclusively in the continuous forest in the Amazon study was 25% (similar to the average value of our two phases: 24%), and in the Araucaria study it was 36%. In the Amazon, this value for secondary forests was 8%, and for eucalyptus plantations 11% (similar to the values of 9% for secondary forests and 12.5% for eucalyptus plantations in our study). In the Araucaria study, the value for eucalyptus plantation was 45%. In the Amazon, the average proportion of the species present in the forest that were present in secondary forests (59%) was higher than in eucalyptus plantations (47%); a similar pattern was observed in our study, though average values were higher (74% and 61%, respectively). In the Araucaria study, eucalyptus plantations maintained only 34% of the species found in the Araucaria Forest. The structure of all the 15 communities differed significantly among habitats in the study of Barlow and collaborators (Barlow et al. 2007), whereas in our study we detected differences in 8 out of 11 communities studied (structure comparisons were not performed in the Araucaria study).

Considering this comparison and also that the forest cover in the landscape studied in the Amazon was much larger than in our study, it would be possible to suggest that in fragmented environments of the Atlantic Forest in southern Bahia, the importance of eucalyptus monocultures for the conservation of forest species may be slightly higher than in the Amazon. However, the eucalyptus areas in the study of Barlow and collaborators (Barlow et al. 2007) were sampled between 4 and 6 years after planting, whereas in our study sampling was carried out 7 years after planting, thus there was more time for colonization by forest populations (Lindenmayer and Hobbs 2004). Moreover, the management regime of the eucalyptus monoculture studied in the Amazon includes periodic removal of the understory with herbicides, whereas in the Atlantic Forest area studied by us the understory was not removed and contributed to the increase in heterogeneity and resources offer to the fauna (Lindenmayer and Hobbs 2004).

The results from Fonseca and collaborators for the Araucaria Forest (Fonseca et al. 2009) seem to go against this suggestion

that the monoculture management could influence its ability to maintain forest species. The stands of eucalyptus areas in this study were 8, 14 and 30 years old when sampled and, according to the authors, “most stands hold a close and relatively complex understory, due to the ecologically-sustainable practices adopted by the National Forest” (Fonseca et al. 2009). Therefore, it could be argued that only the management regimen of the monocultures (related to age and the development of understory) would not be able to explain well the percentage of forest species present in the eucalyptus plantations of the three studies compared here (and therefore the conservation values of these stands). However, the Araucaria study was the only one to include fruit-body producing fungi and epiphytic angiosperms among the sampled communities. These groups presented the highest number of species in the Araucaria Forest (142 and 37 respectively), of which 77 and 29 were found exclusively in the forest. Moreover, the percentage of these forest species found in the eucalyptus plantations was less than 20% for the fungi and less than 10% for the epiphytic angiosperms. The inability of these forest species to colonize the eucalyptus plantations and their numeric relevance to the total richness of the forest in the study could explain the discrepancy of the results of this study when compared to those from Amazon and Atlantic Forest from southern Bahia.

Likewise the studies in the Amazon and Araucaria Forest, our results point out the importance of large remnants of primary forest or forests at advanced stage of regeneration for the conservation of forest species, since from one quarter to about half of all species present in the studies were found only in forests. Another side of this importance is the potential role of continuous forests and large forest remnants for providing organisms that colonize small forest remnants. Evidence of this importance has been obtained both in studies in the Amazon (Gascon et al. 1999) and in anthropogenic Atlantic Forest landscapes in southern Bahia, which comprise cacao plantations in shade forest systems (‘cabruças’) and regenerating forest fragments (Pardini et al. 2002).

The small regenerating remnants assessed in the present study were able to harbor a large portion of the forest fauna, similarly to what has been observed in secondary forests in the Amazon (Barlow et al. 2007). The passive restoration of degraded areas (i.e., natural colonization and secondary succession (Rey Benayas et al. 2003)) is frequently the only regeneration process used in tropical systems due to its low cost (Aide et al. 2000), and the capacity of these environments to harbor a significant part of the forest fauna reinforces their importance for biodiversity conservation. However, the restoration of the original species richness in secondary forests can take from 20 to 40 years, and the restoration of the community structure seems to be an even slower process (Aide et al. 2000; Dunn 2004) that depends on the proximity to source areas (Redi et al. 2005). A study on lizards carried out in the Atlantic Forest of northern Bahia, for example, detected that 28 years of passive restoration were not enough for a lizard community to be completely recomposed in a secondary fragment of 567 ha, despite its proximity to a source area (1,390 ha) that harbored populations of forest species (Guer-

ro and Rocha 2010). This finding suggests that active reintroduction can be necessary to accelerate or complete the process of faunal restoration in secondary forests.

In comparison to regenerating forests, the eucalyptus plantations harbored a lower proportion of the fauna present in forests, both in the study carried out in the Amazon (Barlow et al. 2007) and in our study. Such differences, however, are small and based on numerical comparison of percentages. Moreover, the difference in our study is influenced mainly by the forest anuran species, which was much less rich in the eucalyptus plantations. Therefore, the capacity of the monoculture to harbor forest species could be seen, in our study, as moderate and almost similar to that of regenerating forests. On the other hand, such fragments were more able to maintain the relative abundances of the species similar to those found in the reference forest, and therefore could have a higher value to conservation. This result could point out an opportunity for increasing connectivity in anthropogenic forest landscapes, in particular when eucalyptus monocultures replace less suitable matrices such as pastures. However, we should also consider that the intermediate capacity of eucalyptus plantations to harbor forest species was observed during an advanced development stage, after seven years of growth and immediately before logging. Hence, an intermediate capacity is the maximum expected, and only at specific times along 7-year cycles (in the last years of eucalyptus growth).

The conservation value of the eucalyptus monoculture in southern Bahia varied among taxa. The percentage of forest species that were found in the monoculture varied from 23% (coleopterans) to 73% (orthopterans and coleopterans) in plantations close to the main forest remnant, and from 33% (anurans) to 90% (bees) in plantations far away from the main remnant and close to smaller forest fragments. Moreover, the structure of three of the 11 communities evaluated (myriapods, bees and butterflies) did not differ between the main remnant and the plantations, but the other ones did. Such differences in the response of different taxa to changes in the environment are known and expected (Fonseca et al. 2009) and references therein]. Although some contradictory results in the literature may be related to differences in the metrics used to represent the community, it is also known that some taxa may present idiosyncratic responses to changes in the environment due, for example, to their dispersion abilities and habitat specialization (Barlow et al. 2007; Fonseca et al. 2009).

A recent review on fauna conservation in large coniferous forests and eucalyptus plantations was carried out in Australia (Lindenmayer and Hobbs 2004), where much more data are available. This review suggests that plantations harbor less diverse communities of vertebrates and invertebrates than native vegetation, as detected in our study. It also concludes that three characteristics affect positively the presence of wild species in plantations: (a) heterogeneity degree at landscape level (e.g., presence of forest remnants and riparian vegetation interspersed with or adjacent to plantations), (b) spatial and temporal logging pattern of the monocultures (e.g. constant maintenance of mature plantations connecting remnants; avoidance of synchronic logging in extensive areas), and (c) increase in structural complexity inside

monocultures (e.g. maintenance of native species, stumps of older trees inside plantations).

As discussed by Lindenmayer & Hobbs (Lindenmayer and Hobbs 2004), there must be a conflict of interest between production and biodiversity conservation in monocultures created in forest landscapes. The eucalyptus monocultures of southern Bahia are among the most productive of the world and represent one of the most profitable activities in Brazil (SBS 2007). On the other hand, they are located in one of the most important areas for biodiversity conservation in the planet. The development of monoculture management technologies and landscape planning that conciliate profit with biodiversity conservation must be the focus of interactions between government, research centers and companies that plan the development of the region.

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