

Effects of forest fragmentation on dipterofauna (Calliphoridae) at the Reserva Biológica do Tinguá, Nova Iguaçu, RJ

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(With 3 figures)

Abstract

Samples were collected every month in three different sites of the Reserva Biológica do Tinguá, Brazil: site A was located on the border of the forest and sites B and C were located 1,000 and 500 m, respectively, towards the forest interior. The objective was to determine edge effects on a fragment of the Atlantic Forest. The greatest species richness was observed in sites A and B (23 species), compared with site C (16 species). Site A showed the greatest abundance and constancy, independent of the degree of synanthropy. Asynanthropic species were more abundant and constant in sites B and C. Site B showed the greatest diversity; and sites A and B showed the greatest similarity of populations. There was no significant correlation between Calliphoridae richness and canopy openness except in site C. Richness and abundance were positively correlated with subwood density, except for richness in site B.

Keywords: edge effect, diversity, constancy index.

Efeitos da fragmentação florestal sobre a dipterofauna (Calliphoridae) na Reserva Biológica do Tinguá, Nova Iguaçu, RJ

Resumo

Foram realizadas coletas mensais em três pontos da Reserva Biológica do Tinguá: o ponto A estava localizado na borda da mata, e os pontos B e C estavam localizados, respectivamente, a 1000 e 500 m no interior da mata. Objetivou-se determinar os efeitos de borda em um fragmento da Mata Atlântica. A riqueza de espécies foi maior nos pontos A e B (23 espécies), em relação ao ponto C (16 espécies). No ponto A, ocorreu maior abundância e constância de califorídeos, independente do seu grau de sinantropia. Nos pontos B e C, as espécies assinantrópicas foram mais abundantes e constantes. O ponto B apresentou a maior diversidade registrada. Os pontos A e B apresentam maior similaridade quanto às populações. Não houve significativa correlação entre riqueza de califorídeos e abertura do dossel, exceto no ponto C. Riqueza e abundância correlacionaram-se positivamente com densidade do sub-bosque, exceto para riqueza no ponto B.

Palavras-chave: efeitos de borda, diversidade, índice de constância.

1. Introduction

Habitat fragmentation drastically affects forest ecosystems, creating different microenvironments on the edge and inside the fragment producing an abrupt transition between the forest and the habitat around it. The most important consequences of fragmentation are the reduction of the area available for habitats and the increased levels of exposure to light, variations in temperature and wind (Bierregaard et al., 1992; Rodrigues, 1998). These edge effects are sometimes evident up

to 500 m towards the forest interior (Laurance, 1991). However, they are frequently more intense in the first 35 m (Rodrigues, 1998).

The width edge effects may vary among forest fragments, due to different aspects, such as microclimate, tree species composition, and plant density. Width estimations should take into account the possibility that edge effects may be more intense at a certain distance

from the edge than on the edge of the fragment itself (Rodrigues, 1998).

Many species may be eliminated by forest fragmentation. The loss of area may immediately exclude some species, if they are rare and found in specific spots. Habitat fragmentation may also divide one large population into subpopulations located in small areas, making them more vulnerable to endogamic depression, genetic changes, and increasing their chance of extinction. These changes may enable the proliferation of species adapted to the new environmental conditions, where these would compete with the original species found in the area (Laurance et al., 2001).

Invertebrate populations are potential indicators of environment quality in fragmented habitats due to their short life cycle and low resistance to environmental imbalances (Brown and Hutchings, 1997).

Forest degradation is more pronounced in the tropics because of growing deforestation rates. However, because of the great biodiversity and complex biotic interactions found in tropical forests, concerns should be even greater than they are now (Didham et al., 1996). The Atlantic Forest is considered to be an area of extraordinary diversity and is known to concentrate a large number of endemic species, endangered due to the rapid and significant loss of habitat (Myers et al., 2000).

Human effects in the Reserva Biológica do Tinguá have previously been reported by Marinho et al. (2006), and this analysis drew attention to the need for further studies in the area. The parameters of interest include an estimation of forest conservation and human effect on dipterans in the Calliphoridae family; predominance and diversity of species on the edge and forest interior; better understanding of population dynamics; current dispersion of exotic species; identification of bioindicator species and, of paramount importance, in order to support preservation and conservation of the Atlantic Forest.

The objectives of this study were to determine border effects on the richness and abundance of Calliphoridae species found in artificial edges and preserved forest interiors. The study also aimed to identify resident and migrant species; to estimate the similarity between populations, as well as their diversity; to correlate subwood density and canopy openness with abundance and richness; and to group the species found as a function of their habitat.

2. Materials and Methods

The Reserva Biológica do Tinguá is a protected area of approximately 26,000 ha that presents a very diverse fauna, with large mammals, such as the mountain lion (*Puma concolor* Linnaeus, 1771) and other endangered species of the Atlantic Forest; as well as flora including an abundance of *jequitibás* (*Cariniana* sp. Casar, 1842), *jatobás* (*Hymenaea courbaril* Linnaeus, 1753) and orchids (Orchidaceae). The area provides water for part of the city of Rio de Janeiro and the Baixada Fluminense. Its

main access (22° 58.559' S and 43° 43.809' W), as well as its widest area, is located in the Baixada Fluminense range (JBRJ, 2002; Braz et al., 2004). It has two deforested corridors that potentially submit this fragment to edge effects. Two subterranean ducts cross these corridors: Orbel 1 (PUC-RIO, 2006) and old *Estrada do Comércio*.

Insects were captured every month, from June 2006 to May 2007, using black traps according to the protocol proposed by Mello et al. (2007) (Ferraz and Aguiar-Coelho, 2008). Three sites of capture were determined on the edge and the forest interior based on the analysis of Laurance (2000), in the Amazon, who considered that edge effects in plant communities may range from 400 m to several kilometres towards the interior of the forest. Thus, one of the sites chosen was located on the edge, and the others more than 500 m away from it.

The bait comprised 400 g of sardines thawed for 24 hours at refrigeration temperatures. Two traps were placed in each site, 5 metres apart and 1.5 m high (Neto et al., 1995). Canopy openness was determined using a manual forest densiometer (Guilherme, 2000). Density of the subwood (mean number of medium-sized plants located in a one-metre radius of each trap) was also recorded.

Site A (22° 58.788' S and 43° 43.459' W) was located 500 metres from the entrance of the Reserve, showing mean canopy openness equal to 32% and subwood density equal to 12.25 (ind.m⁻¹). This site exhibited cactuses (Cactaceae), jackfruit (*Artocarpus heterophyllus* Lamarck) and bromeliads (Bromeliaceae). Site B (22° 58.523' S and 43° 44.540' W) was located 1,200 m away from the entrance of the Reserve, on the *Estrada do Comércio*, and 1,000 m towards the interior of the forest. This site displayed primarily bamboo (Bambusoideae), mean canopy openness equal to 24.50%

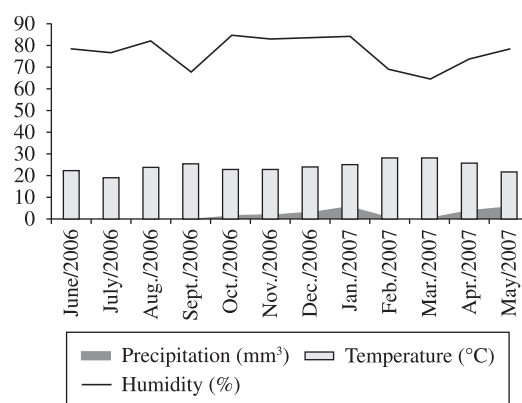


Figure 1. Climatic variables from June 2006 to May 2007, obtained in the Estação Experimental de Itaguaí/PESAGRO-RIO, Seropédica – Rio de Janeiro (22° 45' S and 43° 41' W); mean temperature (°C), relative humidity (%) and total precipitation during the three days the traps were exposed in each month of collection.

and subwood density of 16.25. Site C (22° 58.350' S and 43° 44.678' W) was located 1,700 m away from the entrance of the Reserve, on the *Estrada do Comércio*, 500 m towards the forest interior. The presence of bromeliads was extensive; mean canopy openness was equal to 21.50% and subwood density was equal to 7.00.

After 48 hours, traps were removed and the insects therein were taken to the *Laboratório de Estudo de Dípteros* (LED) at the *Departamento de Microbiologia e Parasitologia* in *Universidade Federal do Estado do Rio de Janeiro* (UNIRIO). Samples remained frozen until submitted to the identification protocol proposed by Mello (2003).

ANOVA was used to compare insect richness among the areas, followed by Tukey's test (SYSTAT software). The resident and migrant species in each site were determined by calculating a constancy index. Similarity among Calliphoridae populations in the different sites was determined by Jaccard's coefficient (Dajoz, 1983). Diversity in each site was estimated by the Shannon Index (Rodrigues, 2004). In order to prove the null hypothesis that samples from sites A, B and C were similar, we followed the protocol suggested by Hutchesson (1970) (cited by Zar 1999). A weighted diversity index (Hp) was calculated for each sample, as a function of the frequency of each species: $H_p = (N \log N) - (\sum f_i \log f_i)/N$, where f_i = frequency

(number of individuals) recorded for species i . Variance of the weighted diversity index was: $\text{var} = [\sum f_i \log^2 f_i - (\sum f_i \log f_i)^2] / N/N^2$. The difference between variances obtained for each sample was calculated: $D_{\text{var}} = \sqrt{(\text{var}_1 + \text{var}_2)}$, and the t value was obtained: $t = H_{p_1} - H_{p_2} / D_{\text{var}}$. The degree of freedom associated with the t value was calculated: $g.l. = (\text{var}_1 + \text{var}_2)^2 / (\text{var}_1^2 / N_1) + (\text{var}_2^2 / N_2)$. After this, the t value calculated was compared with t values from a table.

Species observed in the three sites were also compared using quantitative cluster analysis in order to assess if grouping was a function of habitat type (Zar, 1999); we used Euclidean metrics as a way to measure the distance (STATISTICA software, 1999 edition). Pearson's correlation coefficient was used to assess the correlation between subwood density and canopy openness with abundance or richness in each of the sites.

Data on temperature, relative humidity and precipitation were obtained in the *Estação Experimental de Itaguaí/PESAGRO-RIO*, Seropédica- RJ (22° 45' and 43° 41' W) (Figure 1).

3. Results

A total of 8,516 Calliphoridae individuals from 11 genera and 26 species were captured (as shown in Table 1). Figure 2 shows the nine most abundant species.

Table 1. Absolute number of individuals and relative frequency (f) of Calliphoridae species, captured in three different sites at Reserva Biológica do Tinguá, RJ, in three sites, from June 2006 to May 2007.

Species	Site*			Total	f (%)
	A	B	C		
<i>Calliphora vicina</i>	0	1	0	1	0.01
<i>Chloroprocta idioidea</i>	18	3	8	29	0.34
<i>Chrysomya albiceps</i>	1161	256	67	1484	17.43
<i>Chrysomya megacephala</i>	729	49	40	818	9.61
<i>Chrysomya putoria</i>	10	2	0	12	0.14
<i>Cochliomyia hominivorax</i>	7	2	1	10	0.12
<i>Cochliomyia macellaria</i>	59	13	0	72	0.85
<i>Eumesebrinella besnoiti</i>	3	1	0	4	0.05
<i>Eumesebrinella pauciseta</i>	13	52	10	75	0.88
<i>Eumesebrinella quadrilineata</i>	0	2	4	6	0.07
<i>Eumesebrinella randa</i>	0	2	1	3	0.04
<i>Hemilucilia segmentaria</i>	147	122	29	298	3.50
<i>Hemilucilia semidiaphana</i>	1045	572	195	1812	21.28
<i>Huascaromusca aeneiventris</i>	8	24	0	32	0.38
<i>Huascaromusca purpurata</i>	1	1	0	2	0.02
<i>Laneela nigripes</i>	302	479	860	1641	19.27
<i>Mesembrinella bellardiana</i>	214	874	293	1381	16.22
<i>Mesembrinella bicolor</i>	10	17	11	38	0.45
<i>Mesembrinella semihyalina</i>	51	109	70	230	2.70
<i>Paralucilia borgmeieri</i>	2	0	0	2	0.02
<i>Paralucilia fulvinota</i>	1	0	0	1	0.01
<i>Paralucilia nigrofacialis</i>	3	1	2	6	0.07
<i>Paralucilia paraense</i>	1	0	0	1	0.01
<i>Paralucilia pseudo-lyrcea</i>	34	12	0	46	0.54
<i>Lucilia eximia</i>	62	331	86	479	5.62
<i>Lucilia sericata</i>	9	19	5	33	0.39
Total	3890	2944	1682	8516	-

*Site A, located 500 m from the entrance of the Reserve (edge); Site B and Site C, located respectively 1,000 and 500 m towards the interior of the forest;

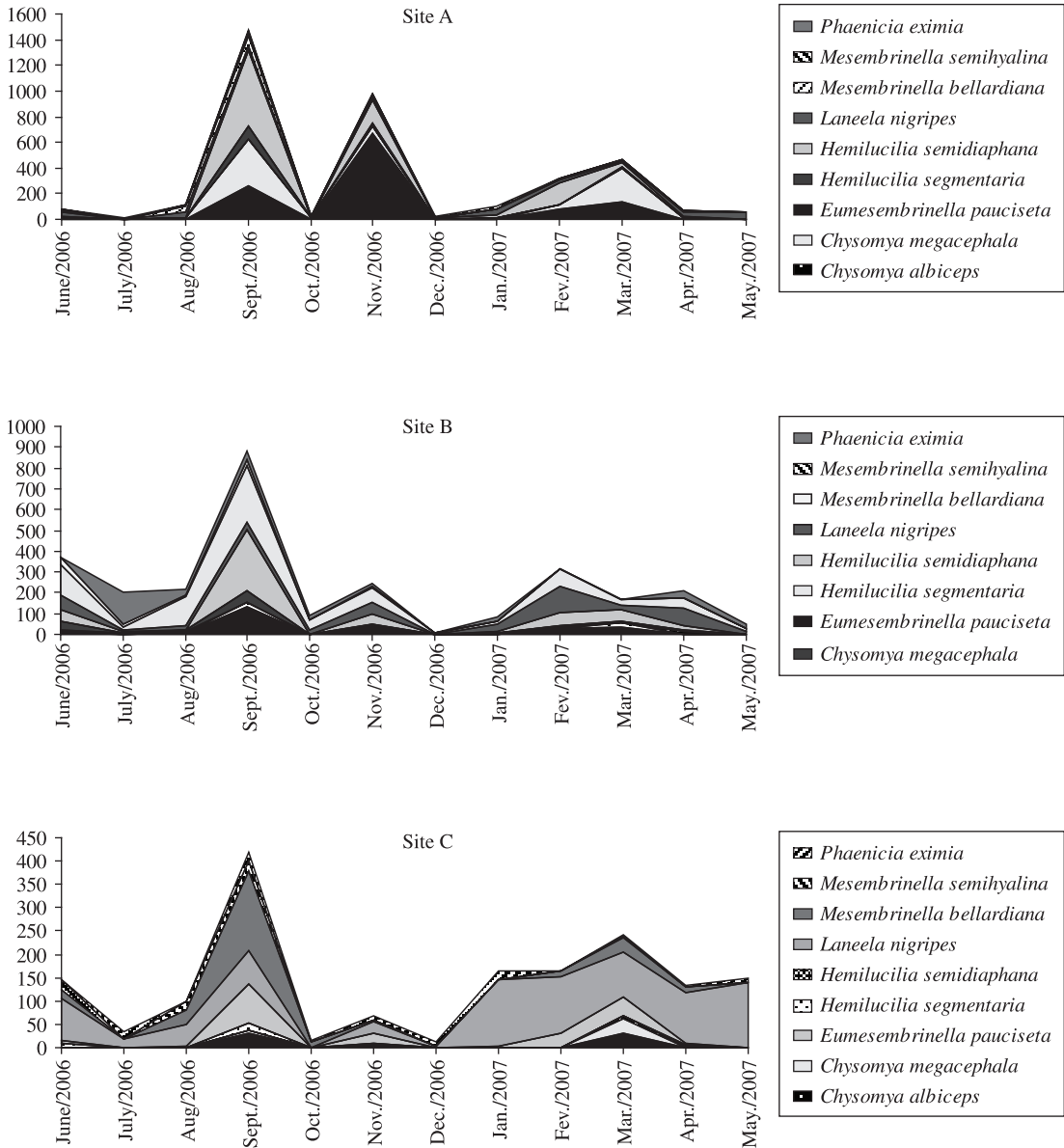


Figure 2. The nine most abundant Calliphoridae species collected in the Reserva Biológica do Tinguá from June 2006 to May 2007, in the different collection sites: Site A, located 500 m from the entrance of the Reserve (edge); Site B and Site C, located respectively 1,000 and 500 m towards the interior of the forest.

Species richness did not vary among the different sites ($F = 3,112$, $DF = 2$, $P = 0.058$). Paired analysis of richness in the different months showed that species collected in sites A and B were not different ($P = 0.694$). However, sites A and B were significantly different from site C ($P = 0.001$ and $P = 0.002$, respectively). The three most abundant species were: site A) *Chrysomya albiceps* (Wiedemann, 1819), *Hemilucilia semidiaphana* (Rondani, 1850) and *Chysomya megacephala* (Fabricius, 1794); site B) *Mesembrinella bellardiana* (Aldrich, 1922), *H. semidiaphana* and *Laneela nigripes* (Guimarães, 1977); and site C) *L. nigripes*, *M. bellardiana* and *H. semidiaphana*.

The following species were considered to be constant: *H. semidiaphana*, *L. nigripes*, *M. bellardiana* and *Lucilia eximia* (Wiedemann, 1819). Only *Mesembrinella bicolor* (Fabricius, 1805) was accessory in the three sites. *Paralucilia nigrofacialis* (Mello, 1969) and *Lucilia sericata* (Meigen, 1826) were accidental species. *C. megacephala* was classified as constant only in site A. *Chrysomya putoria* (Wiedemann, 1818), *Cochliomyia hominivorax* (Coquerel, 1858), *Cochliomyia macellaria* (Fabricius, 1775) and *Huascaromusca aeneiventris* (Wiedemann, 1830) were classified as accessory; *Paralucilia borgmeieri* (Mello, 1969) and *Paralucilia paraense* (Mello, 1969) were considered to be accidental.

Mesembrinella semihyalina (Mello, 1967) was the only species exclusively constant at site B; *C. megacephala* was classified as accessory only in this site. *Calliphora vicina* (Robineau-Desvoidy, 1830), *Chloroprocta idioidea* (Robineau-Desvoidy, 1830), *C. macellaria* and *C. putoria* were considered accidental only in site B. None of the species were constant or accidental only in site C, but *C. albiceps*, *Eumesebrinella pauciseta* (Aldrich, 1922) and *Hemilucilia segmentaria* (Fabricius, 1805) were considered accessory only in this site. The similarity coefficient showed greater correspondence between the populations in sites A and B (0.76), followed by B and C (0.53), and finally A and C (0.44).

Shannon Diversity indexes calculated for each site showed that site B was the most diverse (2.01 nats.ind⁻¹), followed by site A (1.87 nats.ind⁻¹), and site C (1.61 nats.ind⁻¹). Total diversity was equal to 2.08 nats.ind⁻¹. The null hypothesis that the samples (sites A, B and C, as measured by the Shannon Index) were similar was proven false (according to Hutcheson, 1970, cited by Zar, 1999).

The dendrogram (Figure 3) showed similarity between the ten most frequently collected species. The following cluster patterns were observed: *L. nigripes*, separated from the rest, was the main species in site C; *M. bellardiana* showed high abundance in all sites, but mainly in site B; *C. albiceps*, *H. semidiaphana* and *C. megacephala*, the most frequent species found in site A, formed a cluster; another cluster was formed by the species abundant in site B (*E. pauciseta*, *H. segmentaria*, *L. eximia*, *M. bicolor*, *M. semihyalina*).

In site A, subwood density showed significant correlation and inverse with canopy openness ($r = -0.652$; $P = 0.034$), abundance ($r = 0.860$; $P = 0.014$) and richness ($r = 0.850$; $P = 0.015$). Canopy openness did not show significant correlation with abundance ($r = -0.289$; $P = 0.361$) or richness ($r = -0.167$; $P = 0.603$). In site B, subwood density was significantly correlated with canopy openness ($r = 0.583$; $P = 0.041$) and abundance ($r = 0.567$; $P = 0.043$), but not with richness ($r = -0.170$; $P = 0.830$). Canopy openness was not correlated with abundance ($r = 0.374$; $P = 0.231$) or richness ($r = 0.415$; $P = 0.179$) in this site. In site C, subwood density was significantly and inversely correlated with canopy openness ($r = -0.667$; $P = 0.033$) and positively correlated with abundance ($r = 0.752$; $P = 0.024$) and richness ($r = 0.795$; $P = 0.020$). Canopy openness did not show significant correlation with abundance ($r = -0.254$; $P = 0.426$), but it was significantly and inversely correlated with richness ($r = -0.511$; $P = 0.008$).

4. Discussion

While twenty-six species were collected in this study, Mello et al. (2007) and Marinho et al. (2006), in other locations of the same Reserve, captured 13 and 10 species, respectively. All the species collected by these two authors were also cited in this study. Climate conditions and type of bait used, as well as the model, colour and location of the traps may all have influenced these results.

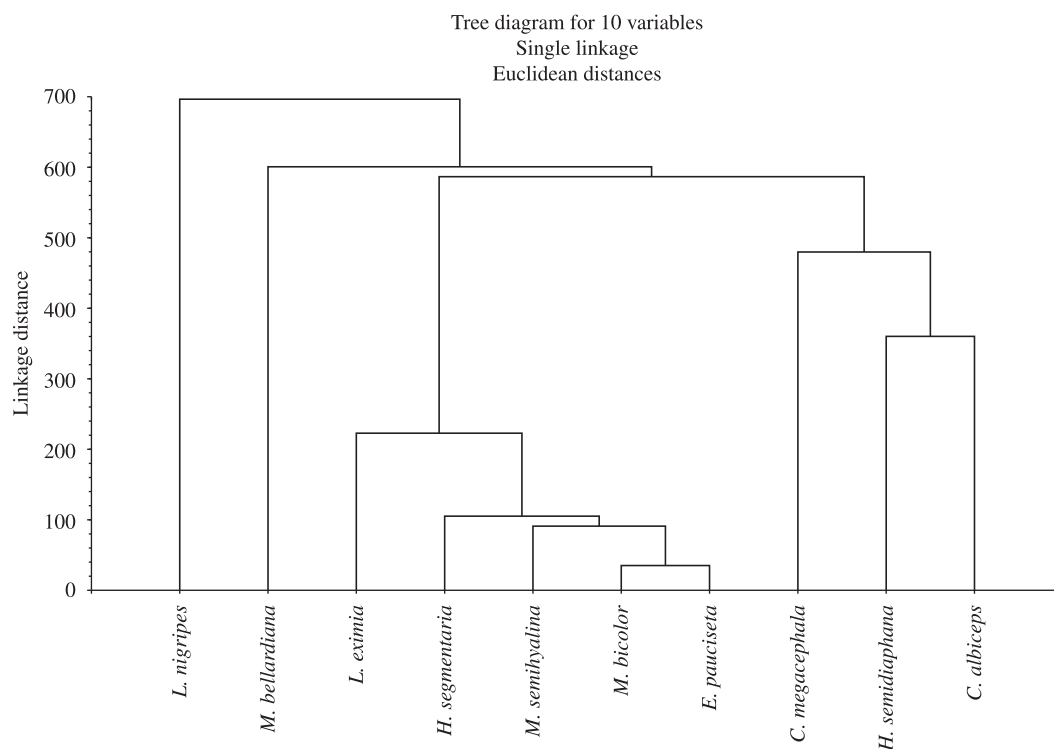


Figure 3. Dendrogram comparing the ten most common species, grouped according to the habitat.

Sites A and B showed the greater richness, aside from being submitted to edge effects, and being located closer to human populations. Laurance and Bierregaard (1997) noted that the richness of birds, primates and several insects decreased as a function of the proximity to the edge, whereas richness of small mammals, amphibians and butterflies increased. Among beetles, richness is influenced by the shape of the fragment, whereas for spiders, the level of isolation of the fragment is the most important factor (Usher et al., 1993).

Although the three sites have significantly different diversity indexes, all of them were very similar when Jaccard's coefficient was considered. High similarity determined by this coefficient may be explained by the proximity of the sites and by the fact that all of them were in the same environment. Seldom does this coefficient reach values over 60% (Mantovani, 1987), and results above 25% are considered to be similar (Mueller-Dombois and Ellenberg, 1974).

The largest canopy openness in Site A, indicates the presence of few high trees in the site, and possibly revealing an area of deforestation. This site probably suffers the greatest influence of abiotic factors, according to the study by Melo (2006), who reported that trees on the edges of the forest showed greater mortality, probably because they are submitted to a rigid increase in abiotic events. However, this author showed that, due to the old age of Orbel 1, the tree community and forest interior are similarly preserved. This fact was supported by other authors, such as Laurance (2000), who suggested that longer fragmentation times minimize impact due to the establishment of protective plant cover, such as vines and bamboos on the edges of forest fragments (Saunders et al., 1991). This protective cover was, in fact, found in site A.

Edge effects are considered to be the main factors affecting fragments of tropical forests. The intensity and rapidity of changes produced by edge effects are influenced by factors such as plant cover, as well as shape, size and age of the fragment (Laurance and Bierregaard, 1997). Differences in the richness and abundance of Calliphoridae specimens observed among the sites studied may be due to edge effects.

The finding of few constant aspects in the Reserva Biológica do Tinguá is probably due to the absence of agricultural practices and to the use of fire in the Reserve. However, its deforesting, constantly maintained corridors show the effects of human actions, such as hunting and selective cutting (Melo, 2006; Rodrigues and Nascimento, 2006), mainly caused by illegal palm heart harvesting (*Euterpe edulis*) (Melo, 2006). Habitat fragmentation increases the access to forest resources and facilitates hunting, wood extraction and agricultural practices (Tabarelli et al., 2004).

Chrysomya megacephala is a highly synanthropic species, as evidenced by the studies by Vianna et al. (2004) in Pelotas, Rio Grande do Sul, and D'Almeida, as well as those performed by D'Almeida and Lopes (1983)

and Guimarães (2006) in Rio de Janeiro. In fact, this species was considered to be exclusively constant in site A, the most affected by human activities. *C. megacephala* is an r-strategist species (Prado and Guimarães, 1982) and responds better to environmental changes, as do generalist and opportunistic species (Didham et al., 1996). This fly may be considered an indicator of the human-modified environment.

C. albiceps that was also extremely abundant and constant, was reported by Ferreira (1978), Linhares (1979), D'Almeida and Lopes (1983) being in human-influenced areas. Both *C. megacephala* and *C. albiceps* were constant in the study by Rodrigues-Guimarães et al. (2004) in a reforestation area in Nova Iguaçu, Rio de Janeiro.

Paralucilia fulvinota (Bigot, 1877), collected only in site A but considered to be an accidental species, is endemic in Brazil and a typical forest inhabitant in neotropical regions (Mariluis and Mulieri, 2003).

Site A exhibited constant species considered by other authors as synanthropic (*C. albiceps*, *C. megacephala*, *L. eximia*) (Ferreira, 1978; D'Almeida and Lopes, 1983) and asynanthropic (*H. semidiaphana* - D'Almeida and Lopes, 1983; Paraluppi, 1996; Guimarães, 2006; *H. segmentaria* - D'Almeida and Lopes, 1983; Ferreira, 1983; Guimarães, 2006; *L. nigripes* - Mello, 2003; *M. bellardiana* - D'Almeida and Lopes, 1983; Marinho et al., 2006; Guimarães, 2006; and *Eumesebrinella* sp. - Guimarães, 2006). This finding demonstrated that although this site was influenced by human actions, it is still suitable for the establishment of forest species. The dendrogram also pointed to the association between asynanthropic and synanthropic species.

In site B, asynanthropic species, such as *M. bellardiana* and *M. semihyalina*, predominated due to the rare presence of humans and a location towards the interior of the forest, which is less affected by climatic variations. However, synanthropic *C. albiceps* was found in this site, showing the great adaptive potential of this exotic species in Brazil.

Site C, the most preserved site, showed the least diversity of Calliphoridae specimens. Site C, as well as sites A and B, showed asynanthropic species such as *H. semidiaphana*, *M. bellardiana*, *L. nigripes*, and synanthropic species, such as *L. eximia*. Other authors reported the movement of *L. eximia* towards the interior of forests and suggested a possible competition with species in the *Chrysomya* genus (Prado and Guimarães, 1982; Ferreira, 1983).

The genus *Eumesebrinella* is only found in forest areas (Mello, 2003), and in this study it was considered accidental in the three sites.

Hemilucilia semidiaphana was constant in the three sites, whereas in the study by Rodrigues-Guimarães et al. (2004) in a reforestation area, the species was considered to be accessory.

A survey in a fragment of the forest in Ilha do Governador, carried out by Leandro and D'Almeida

(2005), also showed *Chloroprocta idioidea* as accidental. These insects were collected at extremely irregular frequencies and the authors observed that the bionomy of the species is still unknown. D'Almeida and Lopes (1983) and Guimarães (2006) considered it to be an asynanthropic species.

Centeno et al. (2004) considered that the decreasing abundance of *Paralucilia pseudo-lyrcea* (Mello, 1969) was an indicator of human actions. This study in Argentina showed this species as an asynanthropic one, of high diversity in natural areas.

Marinho et al. (2006) observed peaks in dipteran capture in May, June, September and January. May, June and September peaks were probably consequences of the presence of *grumixama* fruits (*Eugenia brasiliensis*), while the January peak resulted from guava fructification. Santos (1995) and Azevedo (2001) also observed that the presence of fruit trees influenced dipteran capture. In the present study, the greatest abundance of dipterans occurred in September and November 2006 for the majority of species, which may also be related to the fructification of *grumixama* and jackfruits.

The differences between sites A and C in relation to the Calliphoridae community reflected the differences among the tree community and climatic variables of these sites. Seasonal distribution of Calliphoridae specimens is highly influenced by variations in climatic conditions (Ferreira and Lacerda, 1993).

Subwood density and canopy openness were directly correlated only in site B possibly due to the massive presence of bamboo in this site, showing that the greater the abundance of bamboo (dense subwood), the lower the number of tall trees (larger canopy openness).

Site A showed a significant relationship between subwood density and Calliphoridae abundance and richness. In site B, only subwood density was correlated with abundance. In site C, it was observed that the greater the subwood density, the greater Calliphoridae abundance and richness; larger canopy openness was correlated with less richness. Except in site A, abundance and richness were correlated with vegetation: the denser the subwood, the smaller the canopy openness and the greater the abundance and richness of Calliphoridae. However, in the study by Furusawa and Cassino (2006), the greatest diversity and abundance were observed in sites where edge effects were still observed.

According to Vieira and Mendel (2002), arthropod diversity is related to structural complexity of the habitat. Structurally more complex environments show greater numbers of species because they offer greater availability of habitats, refuge against predators, places for nidification and food resources. Therefore, greater richness and abundance are more likely to occur in denser forests.

Most of these species are highly susceptible to extinction in tropical forests, because they occur in very low densities and show close ecological interactions with other species (Myers, 1987). This explains the low

number of individuals of some species that were collected in this study, such as *C. idioidea*.

Although different results were obtained in the Amazon by Laurance et al. (2002), and in other long-term studies (Debinski and Holt, 2000), no satisfactory concepts have been able to explain edge effects in tropical forests (Rodrigues and Nascimento, 2006). Esposito and Filho (2006) believe that some fly species are more closely associated with pristine environments, whereas others are more connected to affected environments. Therefore, further studies are necessary to determine a new paradigm for understanding these phenomena.

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