

Is the Atlantic Forest protected area network efficient in maintaining viable populations of *Brachyteles hypoxanthus*?

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Abstract Habitat loss and fragmentation are serious threats to biodiversity conservation in the Atlantic Forest. A network of protected areas is essential to the protection of native biodiversity. However, internal and external factors may threaten the preservation of biota, thus population viability analyses (PVA) are important tools in protected area design and management planning. A PVA was carried out, using the computer package VORTEX, to assess the effectiveness of the protected area network within the Atlantic Forest in Brazil in retaining viable populations of the endemic primate *Brachyteles hypoxanthus*. The Brazilian Atlantic Forest has 42 protected areas within *B. hypoxanthus* geographic distribution area, and only five of those were considered to retain viable populations for 50 generations, whereas 28 were predicted to suffer from genetic decay, seven from both genetic decay and demographic stochasticity, and two of them are probably extinct populations. The model indicates that although the protected area network of the Atlantic Forest will likely keep *B. hypoxanthus* populations for the next 50 generations, most of them (35 out of 42, or 83%) will be facing some kind of demographic and/or genetic problem and will probably need management actions to be implemented in order to ensure their persistence.

Keywords *Brachyteles* · Population viability analysis · PVA · Protected areas · Risk assessment · Wildlife management

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Introduction

The Atlantic Forest is one of the world's 34 Biodiversity Hotspots and one of the most devastated and highly threatened ecosystems on the planet (Myers et al. 2000; Mittermeier et al. 2004). Habitat loss and fragmentation have severely altered most of the Atlantic Forest, and only about 7.25% of its original area is left (Câmara 2003; Hirota 2003). To make matters worse, the remnant area is scattered into 10s of 1,000s of small fragments and only a few large ones (Ranta et al. 1998; Gascon et al. 2000). Threats to the biodiversity of the Atlantic Forest are exacerbated by the fact the region is home to approximately 70% of Brazil's 169 million people (Jacobsen 2003; Pinto and Brito 2003). Human population density in the Atlantic Forest has been estimated at 74.5 people/km² (Jacobsen 2003). Furthermore, about 80% of Brazil's gross domestic product is generated in the region that encompasses what remains of the Atlantic Forest, and the region shelters Brazil's largest industrial and silvicultural centers (Pinto and Brito 2003; Young 2003). The Atlantic Forest is a Biodiversity Hotspot where the pace of change is among the fastest and where the need for conservation action is most compelling (Galindo-Leal and Câmara 2003). Loss of habitat in tropical forests is defined by much more than reduction and fragmentation of the original forest, which in the case of the Atlantic Forest has been staggering. Remaining forest habitats are also being drastically transformed by a series of often interrelated processes involving fragmentation, hunting, fire, extraction of forest products and the invasion of alien species (Dean 1995; Galetti and Aleixo 1998; Galetti and Fernandez 1998; Grelle et al. 1999; Cullen et al. 2000; Gascon et al. 2000; Chediack and Baqueiro 2003; Galindo-Leal 2003; Reaser et al. 2003).

Establishing protected areas has been one of the most important tools for conserving some components of biodiversity (Shafer 1999; Bruner et al. 2001), and the number of protected areas that have been created in the Atlantic Forest has risen dramatically over the past 40 years (Galindo-Leal and Câmara 2003; Lairana 2003). Although the number of protected areas in the Atlantic Forest is among the highest in Brazil when compared to other biomes, most of them are small and, in most cases, barely able to maintain viable populations for some species or to hold off the pressures of development (Lairana 2003; Pinto and Brito 2003; Brito and Grelle 2004). However, the establishment of a protected area network does not ensure the permanent preservation of biota (Newmark 1987, 1995, 1996; Grelle and Brito 2006), since a wide range of actions are needed to enhance long-term viability of populations (Shafer 1999).

The objective of the present study is to use population viability analysis as a modeling tool to evaluate the effectiveness of a protected area network for the conservation of viable populations *Brachyteles hypoxanthus*, a Critically Endangered endemic primate of the Atlantic Forest.

Materials and methods

Natural history data of *Brachyteles hypoxanthus*

The muriquis (*Brachyteles*) are the largest Neotropical monkeys, and are endemic to the Atlantic Forest (Aguirre 1971). Traditionally, muriquis have been classified as a monotypic species, *B. arachnoides*. Although a controversial issue for some, new morphological and genetic data suggest that the northern and southern populations of *Brachyteles* can be arranged into separate species: *B. hypoxanthus* and *B. arachnoides* respectively (Coimbra-Filho et al.

1993; Rylands et al. 1995; Brito 2004). *B. hypoxanthus* occurs from southern Bahia to southern of Minas Gerais, but its geographic range is poorly known (Aguirre 1971; Rylands et al. 1998). *B. hypoxanthus* is diurnal and arboreal (Aguirre 1971), but it descends to the ground to cross gaps in the canopy, drink water, eat fallen fruits and, in some cases, to play (Dib et al. 1997). The most important items in the murrelet's diet are fruits, flowers and leaves (Milton 1984; Strier 1991a). Although considered frugivores, some murrelet populations include more than 50% of leaf material in their diet, while also including to a smaller scale, bamboo and bark (Strier 1991a; Carvalho et al. 2004). Murrelets live in large groups of more than 50 individuals (Strier 1993/1994). Males are philopatric whereas females generally disperse into other groups as they reach adolescence at about 6 years of age (Strier 1990, 1991b, 2005; Printes and Strier 1999; Strier and Ziegler 2000). Males first reproduce at 7 years, and females at 9 years (Rylands et al. 1998; Strier and Ziegler 2000; Strier 2005). Females give birth to one young at a time and interbirth interval is estimated at 3 years (Strier 1991b; Strier et al. 2001). Two philopatric females were observed to reproduce at 7.5 (Strier 1991b) and 7.25 years of age (Martins and Strier 2004).

PVA model

The computer simulation package VORTEX version 8.21 was used (Miller and Lacy 1999). VORTEX is a Monte Carlo simulation of the effects of deterministic forces (e.g. trends in carrying capacity) as well as demographic, environmental and genetic stochasticity and catastrophes on the dynamics of wildlife populations (Lacy 1993, 2000a; Miller and Lacy 1999). This package is one of the most often used for PVA focusing endangered populations, including in workshops with officers from conservation and land management agencies (Lindenmayer et al. 1995).

Protected area network effectiveness

It was assumed that *B. hypoxanthus* was originally found in all protected areas within the geographic range of the species (Fig. 1). In addition, it was also assumed that all protected areas were isolated, precluding dispersal. Protected areas were grouped into three size categories: small (<661 ha; <40 individuals), medium (661–11,570 ha; 40–700 individuals) and large (>11,570 ha; >700 individuals) (Brito and Grelle 2006). The dynamics of single, isolated populations were analyzed using a time-frame of 50 generations. Carrying capacities of protected areas were estimated based on published data on population densities for *B. hypoxanthus* (Strier and Fonseca 1996/1997). Five hundred iterations were run for each protected area and the initial population size was set at half its carrying capacity. Demographic parameters used as input to the model were based on previously demographic published data and PVA studies on *B. hypoxanthus* (Strier 1991b, 1993/1994, 2000, 2005; Rylands et al. 1998; Brito and Grelle 2006; Strier et al. 2006; Coutinho 2007), and a summary of the PVA input data set is provided in Appendix A.

Small protected areas are effective if their populations have 99% chance of persistence (Shaffer 1981). These populations probably suffer from genetic and demographic stochasticity. Thus, viability predictions for small protected areas do not predict the likelihood of falling below a quasi-extinction threshold, but rather predict local extinction. Medium protected areas are effective if their populations have <1% chance of falling below the quasi-extinction threshold (40 individuals), but such populations will likely show declines in gene diversity. These protected areas should house demographically viable populations, but that would suffer from the negative effects of genetic erosion, thus seriously reducing

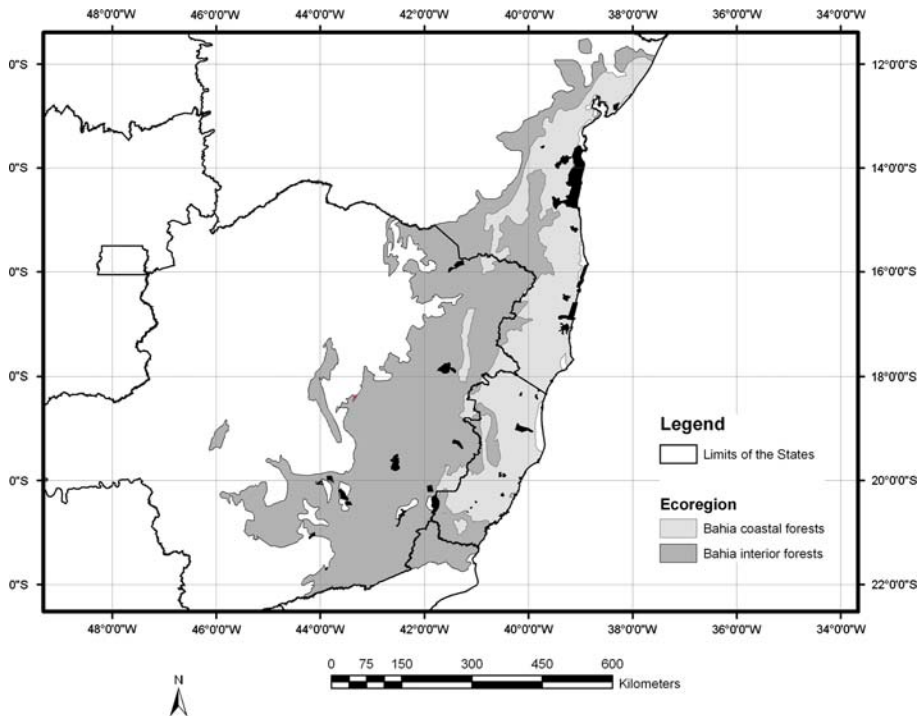


Fig. 1 Protected areas (black polygons) encompassed within the geographic range of the northern muriqui *B. hypoxanthus* in the Atlantic Forest

their capacity for adaptation to environmental changes (Brito and Grelle 2006). Large protected areas are effective if their populations have <1% chance of falling below the quasi-extinction threshold of 700 individuals, and retain at least 90% of their original heterozygosity (Foose et al. 1986; Foose 1993). Such protected areas should retain demographically and genetically viable populations keeping evolutionary potential.

Sensitivity analysis

Sensitivity analysis measures the extent of change in the modeled population output values due to a known change in assumptions. It separates which assumptions the model is particularly sensitive, from the assumptions that are less influential (McCarthy et al. 1995). Sensitivity analysis provides an indication of the impact that errors in assumptions, or real changes due to management or threats, could have on the outcome. The model was examined for sensitivity to variation in inbreeding depression, carrying capacity, mortality rates, sex ratio and percentage of reproductive females. The effect of inbreeding was examined by introducing inbreeding depression to the scenarios. As the actual impact of inbreeding on *B. hypoxanthus* populations is unknown, the standard lethal equivalent median value for juvenile mammal survival of 1.57 per haploid genome was used (Ralls et al. 1988). As VORTEX only models inbreeding impact on juvenile survival, the simulated effect of inbreeding is probably conservative (Lacy 1993). Sensitivity in carrying capacity was evaluated as declines to 50 and 10% of protected area's original carrying capacity after 50

generations, simulating habitat loss. These rates of habitat loss are probably conservative, as estimates of deforestation in the Atlantic Forest are greater than this value (Mata Atlântica SOS 2002; Hirota 2003). Sensitivity to mortality was examined by increasing mortality rate by 5, 10 and 20% (Rylands et al. 1998). Scenarios evaluating sex ratio sensitivity were run using values of 0.500 and 0.650 males/females (Rylands et al. 1998). Regarding percentage of reproductive females, sensitivity analysis was run with scenarios of 20 and 33% of females reproducing at any given year (Rylands et al. 1998). We also assess uncertainty by running scenarios with increased standard deviation values (+5, +10 and +20%).

Results

Protected areas network effectiveness

It is estimated that only five out of 42 protected areas would retain demographic and genetically viable populations (ASPE Córrego Soberbo e Retiro, PN Caparaó, APA Cariaba/Trancoso, PE Rio Doce and ASPE Rio Todos os Santos) (Fig. 2; Table 1). These are optimistic estimates, since we did not evaluate habitat quality inside protected areas, a factor that may affect model predictions. All the area within protected areas was considered to be suitable habitat, however, most probably only a fraction of it offers adequate habitat for *B. hypoxanthus*.

We modeled that from six small protected areas (<40 individuals), four (PE Cachoeira da Fumaça; ASPE Córrego do Taboão, ASPE Adjacentes do PE Rio Doce and APA Praia Mole) had extremely high probabilities of extinction (Table 1), with mean times to extinction (generations \pm SD) estimated, respectively, as: 0.52 ± 0.34 , 2.86 ± 2.76 , 23.86 ± 14.42 . All of the six showed a great loss of heterozygosity, which seriously threat genetic integrity. They may also suffer from the deleterious effects of demographic stochasticity, such as biases in population sex ratio and age structure. We also modeled population trends for 23 medium protected areas (40–700 individuals) (Table 1). Such protected areas and population sizes should offer protection against demographic stochasticity, but not from genetic erosion. However, populations from three such protected areas showed high probabilities of falling below the population quasi-extinction threshold size of 40 individuals and suffer from demographic stochasticity: PE Mata das Flores, EE Caratinga and PE Pedra Azul (Table 1). Large protected areas (>700 individuals) represent the threshold size

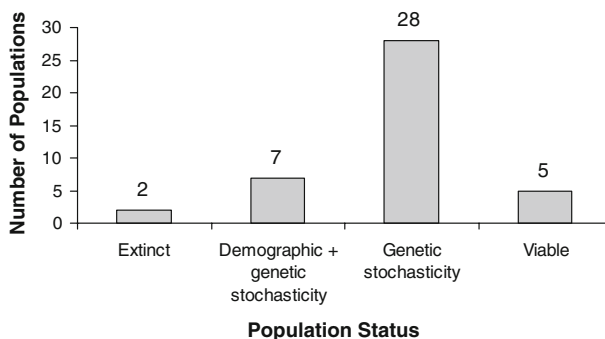


Fig. 2 Status of *B. hypoxanthus* populations in the network of protected areas of the Atlantic Forest

Table 1 Results from modeling the viability of *Brachyteles hypoxanthus* populations within reserves in the Brazilian Atlantic Forest for a time period of 50 generations

Small reserves (<661 ha)	Area (ha)	K	PE	H
<i>PE Cachoeira da Fumaça</i>	27	2	1.0000	–
<i>ASPE Córrego do Taboão</i>	100	6	1.0000	–
<i>ASPE adjacentes PE Rio Doce</i>	300	18	0.3820	0.0243
<i>APA Praia Mole</i>	347	21	0.1640	0.0301
<i>EE Córrego dos Fechos</i>	600	36	0.0000	0.1425
<i>PE Forno Grande</i>	621	38	0.0000	0.1353
Medium reserves (661–11,570 ha)	Area (ha)	K	Q ₄₀	H
<i>PE Mata das Flores</i>	800	48	1.0000	0.2290
<i>EE Caratinga</i>	860	52	1.0000	0.2672
<i>PE Pedra Azul</i>	1,240	75	0.5440	0.3911
<i>PE Ibitipoca</i>	1,488	90	0.0020	0.4781
<i>RB Córrego Grande</i>	1,489	90	0.0000	0.4774
<i>RB Duas Bocas</i>	1,493	90	0.0020	0.4792
<i>PE Paulo César Vinha</i>	1,500	91	0.0000	0.4754
<i>APA Lagoa Encantada</i>	1,800	109	0.0000	0.5290
<i>EEE Nova Esperança</i>	1,939	117	0.0000	0.5774
<i>APA Santo Antônio</i>	2,300	139	0.0000	0.6293
<i>RB Córrego do Veadó</i>	2,383	144	0.0000	0.6267
<i>FN Rio Preto</i>	2,826	171	0.0000	0.6842
<i>RB Augusto Ruschi</i>	3,572	216	0.0000	0.7391
<i>APA Pico do Goiapaba-Açu</i>	3,740	226	0.0000	0.7502
<i>PE Rola Moça</i>	3,940	239	0.0000	0.7653
<i>APA Coroa Vermelha</i>	4,100	248	0.0000	0.7649
<i>APA Serra São José</i>	4,758	288	0.0000	0.7923
<i>PE Rio Corrente</i>	5,065	307	0.0000	0.8098
<i>APA Lagoa do Guanandy</i>	5,242	317	0.0000	0.8085
<i>PE Garcia D'Ávila</i>	7,000	424	0.0000	0.8544
<i>PE Serra do Conduru</i>	7,000	424	0.0000	0.8574
<i>PE Itacolomi</i>	7,542	457	0.0000	0.8662
<i>RB Una</i>	10,641	644	0.0000	0.9055
Large reserves (>11,570 ha)	Area (ha)	K	Q ₇₀₀	H
<i>PN Pau Brasil</i>	11,592	702	1.0000	0.9121
<i>PE Sete Salões</i>	12,521	758	1.0000	0.9176
<i>PE Serra do Brigadeiro</i>	13,210	800	1.0000	0.9209
<i>APA Costa de Itacaré e Serra Grande</i>	14,925	904	1.0000	0.9307
<i>APA Cachoeira das Andorinhas</i>	18,700	1132	0.9980	0.9436
<i>PN do Descobrimento</i>	21,215	1284	0.6260	0.9505
<i>RB Sooretama</i>	22,000	1332	0.4380	0.9525
<i>PN Monte Pascoal</i>	22,406	1356	0.3820	0.9534
<i>ASPE Córrego Soberbo e Retiro</i>	24,171	1463	0.0060	0.9573
<i>PN Caparaó</i>	31,853	1928	0.0000	0.9666
<i>APA Cariaba/Trancoso</i>	31,900	1931	0.0000	0.9673
<i>PE Rio Doce</i>	35,970	2178	0.0000	0.9703
<i>ASPE Rio Todos os Santos</i>	43,449	2630	0.0000	0.9756

Carrying capacities (K) were calculated based on mean *B. hypoxanthus* densities (Strier and Fonseca 1996/1997). Reserves in *italics* are not viable within each size category (PE: probability of extinction, H: expected heterozygosity)

between viability and genetic erosion. There are 13 protected areas equal to or larger than this size within *B. hypoxanthus* geographic range (Table 1). However, eight of these areas have a high probability of declining below the quasi-extinction threshold of 700 individuals and consequently suffer from genetic erosion (Table 1). Although *B. hypoxanthus* populations will most probably persist within these protected areas, they will likely suffer from genetic erosion. The results indicate that 78% of the protected areas will house populations with demographic and/or genetic problems, 10% of the populations are likely to be locally extinct and only 12% of the populations are considered viable (Fig. 2; Table 1).

Sensitivity analysis

Inbreeding scenarios resulted in depressed r and population size, but an increase in H_e for the population threshold size of 40 (Table 2). The basic scenario modeled populations with female-biased sex ratio (0.356). Equal sex ratio scenarios (0.500) depressed r and population size, and increased H_e for both population threshold sizes (40 and 700) (Table 2). Male-biased sex ratios (0.650) also increased extinction probability for populations of 40 animals (Table 2). The scenario with 20% of females breeding resulted in decreased r and final population sizes, and increased H_e for population sizes of 40 and 700 individuals (Table 2). The scenario with greater numbers of reproductive females (33%), on the other way, produced increased r and decreased H_e for both population sizes modeled; whereas an increase in final population size was observed only for the population threshold size of 40 individuals (Table 2). Scenarios evaluating mortality rates showed no significant results for most of the output parameters with the exception of a decrease in H_e with a 10% increase in the mortality rate for the population size of 40 (Table 2). Scenarios simulating habitat loss through the decrease in carrying capacity (50 and 90% habitat loss) resulted in increased probabilities of extinction, decreased final population size and H_e for the threshold size of 40 individuals, and decreased population size and H_e for the population extinction threshold size of 700 individuals (Table 2). The results for the scenarios evaluating uncertainty suggest that the model is fairly robust, and for the lower population size (40 individuals), the increase in uncertainty resulted in a decrease in the expected heterozygosity, but did not affect the demographic viability of the population (Table 2). For the larger population size (700 individuals), increasing uncertainty resulted in smaller population sizes (Table 2).

Discussion

There is a common view that demographic effects are likely to be felt before genetic ones (Soulé 1987; Nunney and Campbell 1993; Caughley 1994; Gilligan et al. 1997). There are studies indicating that the accumulation of mildly deleterious mutations does not pose a significant threat to finite sexual populations for 45–50 generations (Gilligan et al. 1997). Yet, increasing numbers of studies are showing that inbreeding depression can impact population viability to a greater extent, more quickly, and less reversibly than previously supposed (Lande 1994; Frankham 1995; Lacy 1997, 2000b). The risks of stochastic demographic and environmental processes, together with the expected effects of genetic drift and inbreeding, should make small populations of muriquis more vulnerable to extinction than large populations inhabiting undisturbed forests.

B. hypoxanthus is one of the most threatened primates surviving in what remains of the Atlantic Forest of southeastern Brazil (Strier and Fonseca 1996/1997). *B. hypoxanthus* has managed to persist, despite severe habitat disturbance, in part because of their adaptable

Table 2 Sensitivity analysis: changes in assumptions and the resulting effects on the model outcome for the two population sizes estimated as demographic and genetic quasi-extinction thresholds for *Brachyteles hypoxanthus* (N = 40 and N = 700, respectively)^a

Scenarios	Population growth rate (r) [mean (SD)]	Probability of extinction [mean (SE)]	Final population size [mean (SD)]	Expected heterozygosity [mean (SD)]
<i>N</i> = 40				
Basic assumptions	0.0608 (0.0669)	0.0000 (0.0000)	39.41 (1.86)	0.1970 (0.2104)
Inbreeding	0.0400 (0.0607)**	0.0000 (0.0000) ^{ns}	38.97 (2.22)**	0.2320 (0.2186)**
+5% Mortality	0.0602 (0.0673) ^{ns}	0.0000 (0.0000) ^{ns}	39.40 (1.96) ^{ns}	0.1850 (0.2095) ^{ns}
+10% Mortality	0.0593 (0.0677) ^{ns}	0.0020 (0.0020) ^{ns}	39.33 (2.72) ^{ns}	0.1736 (0.2116)*
+20% Mortality	0.0579 (0.0683) ^{ns}	0.0000 (0.0000) ^{ns}	39.39 (2.06) ^{ns}	0.1800 (0.2083) ^{ns}
0.500 Sex ratio	0.0465 (0.0629)**	0.0000 (0.0000) ^{ns}	38.98 (2.14)**	0.2561 (0.2239)**
0.650 Sex ratio	0.0269 (0.0589)**	0.0120 (0.0049)*	37.97 (5.26)**	0.2527 (0.2198)**
20% Females breeding	0.0454 (0.0701)**	0.0000 (0.0000) ^{ns}	39.02 (2.40)**	0.2229 (0.2149)*
33% Females breeding	0.0737 (0.0658)**	0.0000 (0.0000) ^{ns}	39.65 (1.84)**	0.1579 (0.2020)**
10% Carrying capacity	0.0581 (0.0823) ^{ns}	0.9760 (0.0068)**	0.23 (0.76)**	0.0370 (0.1283)**
50% Carrying capacity	0.0602 (0.0729) ^{ns}	0.0480 (0.0096)**	18.41 (4.26)**	0.0947 (0.1711)**
+5% Uncertainty (SD)	0.0608 (0.0695) ^{ns}	0.0000 (0.0000) ^{ns}	39.53 (1.86) ^{ns}	0.1837 (0.2085) ^{ns}
+10% Uncertainty (SD)	0.0608 (0.0714) ^{ns}	0.0000 (0.0000) ^{ns}	39.25 (2.14) ^{ns}	0.1775 (0.2071)*
+20% Uncertainty (SD)	0.0608 (0.0736) ^{ns}	0.0000 (0.0000) ^{ns}	39.32 (2.16) ^{ns}	0.1652 (0.2039)**
<i>N</i> = 700				
Basic assumptions	0.0622 (0.0496)	0.0000 (0.0000)	698.38 (8.28)	0.9119 (0.0201)
Inbreeding	0.0597 (0.0487) ^{ns}	0.0000 (0.0000) ^{ns}	698.26 (9.20) ^{ns}	0.9119 (0.0214) ^{ns}
+5% Mortality	0.0614 (0.0500) ^{ns}	0.0000 (0.0000) ^{ns}	698.04 (9.23) ^{ns}	0.9110 (0.0208) ^{ns}
+10% Mortality	0.0606 (0.0503) ^{ns}	0.0000 (0.0000) ^{ns}	698.12 (8.70) ^{ns}	0.9115 (0.0201) ^{ns}
+20% Mortality	0.0590 (0.0508) ^{ns}	0.0000 (0.0000) ^{ns}	697.92 (8.69) ^{ns}	0.9102 (0.0219) ^{ns}
0.500 Sex ratio	0.0483 (0.0453)**	0.0000 (0.0000) ^{ns}	697.56 (8.92)*	0.9271 (0.0145)**
0.650 Sex ratio	0.0296 (0.0395)**	0.0000 (0.0000) ^{ns}	693.59 (13.03)**	0.9319 (0.0139)**
20% Females breeding	0.0464 (0.0552)**	0.0000 (0.0000) ^{ns}	695.14 (12.53)**	0.9179 (0.0181)**
33% Females breeding	0.0747 (0.0475)**	0.0000 (0.0000) ^{ns}	698.85 (8.40) ^{ns}	0.9052 (0.0214)**
10% Carrying capacity	0.0620 (0.0515) ^{ns}	0.0000 (0.0000) ^{ns}	70.28 (2.62)**	0.7851 (0.0716)**
50% Carrying capacity	0.0621 (0.0502) ^{ns}	0.0000 (0.0000) ^{ns}	349.18 (5.59)**	0.8790 (0.0297)**
+5% Uncertainty (SD)	0.0621 (0.0532) ^{ns}	0.0000 (0.0000) ^{ns}	697.78 (9.61) ^{ns}	0.9099 (0.0248) ^{ns}
+10% Uncertainty (SD)	0.0619 (0.0557)**	0.0000 (0.0000) ^{ns}	696.99 (10.22)**	0.9112 (0.0211) ^{ns}
+20% Uncertainty (SD)	0.0621 (0.0589) ^{ns}	0.0000 (0.0000) ^{ns}	696.92 (11.09)**	0.9111 (0.0189) ^{ns}

^a Significance of the difference in output between the basic and changed models was tested using a Students two-tailed t-test

ns: not significant

* $P < 0.05$

** $P < 0.005$

way of life, which includes their ability to exploit secondary as well as primary forest (Strier 1987, 2000). When considered a single species, there was evidence for high levels of genetic polymorphism and heterozygosity on muriqui populations (Pope 1998). However, since the taxonomic revision splitting muriquis into two species, there is evidence that such variability was due to between-species rather than within-population diversity (Fagundes 2005). Recent genetic studies revealed an absence of gene flow among *B. hypoxanthus* populations and high diversity between populations (Fagundes et al. 2005), reinforcing the isolated nature of the current populations and highlighting the need for increased connectivity (both structural landscape connectivity and/or population management through translocation/reintroduction). Genetic variation in *B. hypoxanthus* populations is among the lowest reported for any primate (Fagundes 2005). Even this small

genetic diversity may quickly deteriorate in the next centuries or 10s of generations (Brito and Grelle 2006), hampering the capacity of *B. hypoxanthus* populations to cope with future environmental changes, and making them more prone to extinction (Van Valen 1973; Brito and Fernandez 2000).

The results of the present study may be conservative, since we did not take into account several potential human impacts on northern muriqui populations. According to Rylands et al. (1998), the major impacts are: (1) coffee planting, (2) cattle grazing, (3) fire for land clearing, (4) small-scale timber harvesting, and (5) firewood harvesting. Hunting may be another source of threat for *B. hypoxanthus* populations we did not take into account in our analysis. For *B. arachnoides* at a protected area in São Paulo state, it is estimated that between 5 and 10 muriquis are harvested each year by hunters (Rylands et al. 1998).

The results presented here, which take into account the chances of the population located within each protected area falling below the quasi-extinction thresholds suggested by Brito and Grelle (2006), raise a warning: although the northern muriqui is likely to persist within the current protected area network, viable populations were found in only 12% of the protected areas modeled. A similar analysis with *Micoureus paraguayanus*, an endemic marsupial of the Atlantic Forest, estimated that only 65% of the protected areas modeled contained viable populations (Brito and Grelle 2004). However, Brito and Grelle (2004) point out that *M. paraguayanus* is a small omnivore mammal, with small home ranges and with the capacity to exploit secondary habitat and travel through harsh matrix habitats, and that the 65% estimate of protected areas housing viable populations was not such a good result as could be first thought. While analyzing the results of the efficiency of the protected areas network for the persistence of *M. paraguayanus* populations, Brito and Grelle (2004) argued that the prospect of persistence for larger mammals could be alarming. Our findings seem to corroborate Brito and Grelle (2004) concern for larger mammals, since only 12% of the protected areas are estimated to contain viable populations for the *B. hypoxanthus*.

To ensure population persistence, management should target actions both at the landscape and population levels. Even if they cannot hold viable populations in the long-term, small protected areas may act as stepping stones for dispersing individuals, mitigating the problems that could threaten completely isolated populations (Caughley 1994; Caughley and Gunn 1996; Brito and Fernandez 2000), and facilitating connectivity among larger protected areas. Considering the highly fragmented landscape of the Atlantic Forest in southeastern Brazil, it would be interesting if such small protected areas constituted viable metapopulations. Therefore, the best scenario would be a landscape comprised of large patches (protected areas containing viable populations), connected by sets of medium and small patches (both as protected areas and private properties), forming viable metapopulations. This configuration should maximize the probability of successful dispersal and provide a viable scenario for the conservation of the northern muriqui. Habitat restoration and increased connectivity (corridors and stepping stones) are recognized as fundamental issues for the persistence of the northern muriqui (Mendes et al. 2005a). Due to the highly fragmented state of the Atlantic Forest, it is important to take into account forest patches on private land, and their role in enhancing connectivity and persistence for northern muriqui populations (e.g. Vieira and Mendes 2005), complementing the existing protected area network. Therefore, taking local communities into consideration in conservation planning and management strategies, in order to successfully include private properties into the landscape management targeting northern muriqui conservation (e.g. Pontual and Boubli 2005). This could diminish small-scale deforestation and ensure the maintenance of medium and small fragments located in private properties within the landscape.

For such a landscape management scenario to be effective, it is essential that it is complemented by management and monitoring at the population scale. Management at the population level should have two main objectives: (1) maintain the large, healthy populations above the quasi-extinction threshold limits, tackling with current and potential threats; and (2) ensure connectivity and successful dispersal for smaller populations, minimizing Allee effects. Solitary females have been found in small forest fragments (Mendes et al. 2005b; Coutinho 2007). They dispersed from their natal groups and, due to the low density resulting from habitat fragmentation, they ended up establishing their home ranges in unoccupied patches (Mendes et al. 2005b; Coutinho 2007). In these cases, translocation and/or reintroduction should be taken into consideration, since these isolated females mean a significant loss affecting both population dynamics and persistence (Mendes et al. 2005a, b; Coutinho 2007).

Currently there are only 12 localities with confirmed populations of *B. hypoxanthus* (Mendes et al. 2005a). However, recent studies resulted in the discovery of new populations (Mendes et al. 2005a). In the last 10 years, the total *B. hypoxanthus* population has increased threefold, and the number of confirmed localities has doubled (Mendes et al. 2005a). Therefore, surveys and censuses, both to discover new populations and to gather better data for known populations are of paramount importance to develop better management strategies for the species (Melo and Dias 2005; Mendes et al. 2005a). The present study shows that there are 42 protected areas within the potential range of the species, and surveys to find new populations should be targeted at these protected sites. At the same time, long-term censuses to gather more data on the existing 12 known populations should be a priority for population monitoring. Surveys, censuses, population management, genetic studies and population viability simulation are among the priority actions listed for the northern muriqui conservation (Mendes et al. 2005a).

It is difficult to assess the actual protection afforded by protected areas because many of them lack the basic apparatus necessary to effectively maintain biodiversity (i.e. management plans, land tenure definition, plant and animal inventories, monitoring and law enforcement). Although a few parks do have effective management mechanisms in place, most are only paper parks (Galindo-Leal and Câmara 2003). Nevertheless, when one considers their chronic underfunding and significant land-use pressures, tropical parks have been surprisingly effective at protecting ecosystems and species (Bruner et al. 2001). However, the results presented here suggest that most of the populations of *B. hypoxanthus* found within protected areas in the Atlantic Forest are likely to show some degree of threat when a longer time period is considered. Shorter-time frames usually used in analyses are in accordance with economic and political issues, and even management practices dealing with species protection, but for long-lived species, longer time frames must be taken into account because they better represent biological aspects of the species studied.

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Appendix A

B. hypoxanthus demographic data used as input values to VORTEX

VORTEX 8.21—simulation of genetic and demographic stochasticity

1 Population(s) simulated for 1,000 years, 500 iterations

Extinction is defined as no animals of one or both sexes.

No inbreeding depression

First age of reproduction for females: 9 for males: 7

Maximum breeding age (senescence): 35

Sex ratio at birth (percent males): 35.600000

Polygynous mating;

100.00% of adult males in the breeding pool.

26.50% of adult females produce litters.

EV in % adult females breeding = 12.40 SD

Of those females producing litters, ...

100.00% of females produce litters of size 1

2.00% mortality of females between ages 0 and 1

EV in % mortality = 1.000000 SD

5.70% mortality of females between ages 1 and 2

EV in % mortality = 2.800000 SD

1.00% mortality of females between ages 2 and 3

EV in % mortality = 1.000000 SD

3.60% mortality of females between ages 3 and 4

EV in % mortality = 1.800000 SD

1.00% mortality of females between ages 4 and 5

EV in % mortality = 1.000000 SD

1.00% mortality of females between ages 5 and 6

EV in % mortality = 1.000000 SD

1.00% mortality of females between ages 6 and 7

EV in % mortality = 1.000000 SD

1.00% mortality of females between ages 7 and 8

EV in % mortality = 1.000000 SD

1.00% mortality of females between ages 8 and 9

EV in % mortality = 1.000000 SD

1.00% mortality of adult females ($9 \leq \text{age} \leq 35$)

EV in % mortality = 1.000000 SD

4.80% mortality of males between ages 0 and 1

EV in % mortality = 2.400000 SD

5.60% mortality of males between ages 1 and 2

EV in % mortality = 2.800000 SD

11.80% mortality of males between ages 2 and 3

EV in % mortality = 5.900000 SD

1.00% mortality of males between ages 3 and 4

EV in % mortality = 1.000000 SD

1.00% mortality of males between ages 4 and 5

EV in % mortality = 1.000000 SD

1.00% mortality of males between ages 5 and 6

EV in % mortality = 1.000000 SD

1.00% mortality of males between ages 6 and 7

EV in % mortality = 1.000000 SD

1.52% mortality of adult males ($7 \leq \text{age} \leq 35$)

EV in % mortality = 1.500000 SD

EVs may be adjusted to closest values possible for binomial distribution.

EV in reproduction and mortality will be concordant.

EV in Carrying capacity = 0.00 SD

Deterministic population growth rate (based on females, with assumptions of no limitation of mates, no density dependence, and no inbreeding depression):

$r = 0.063$ $\lambda = 1.065$ $R_0 = 3.402$

Generation time for: females = 19.57 males = 17.87

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