UNIVERSIDADE ESTADUAL DE SANTA CRUZ

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EMBASAMENTO TEÓRICO PARA O MANEJO POPULACIONAL DO MURIQUI-DO-NORTE, *Brachyteles hypoxanthus* (KUHL 1820) (PRIMATES, ATELIDAE)

ANDERSON AIRES EDUARDO

ILHÉUS – BAHIA – BRASIL

Fevereiro de 2010

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Dissertação apresentada à Universidade Estadual de Santa Cruz, como parte das exigências para obtenção do título de Mestre em Ecologia e Conservação da Biodiversidade.

Área de concentração: Ecologia

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Dr. Leonardo Oliveira
(UFRJ e IESB)

Prof.ª Dr.ª Débora Faria
(UESC)

Prof. Dr. Júlio Baumgartem
(UESC)

Prof. Dr. Daniel Brito
(UFG - Orientador)

Dedico este trabalho aos

meus pais, Benedito e Salete,

e à minha noiva Samanta, por todo o apoio,

amor e dedicação.

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À Deus.

“We consider good theoretical work to be akin to impressionism. Rather than interpreting our mathematical constructions as literal depictions, we see them as metaphors for more complex underlying details. It is for this reason that we allow our models to relax some of the literalist constraints in pursuit of a more valuable albeit imperfect impression (...) what results is an image that is less sharp but more meaningful. It is our judgment that this approach will achieve more with less – theorists will construct more valuable pictures out of the limited available data.”

Trecho da tese de doutorado de **Christopher J. Jensen (2007)**

ÍNDICE

EXTRATO

EDUARDO, Anderson Aires, M.S., Universidade Estadual de Santa Cruz, Ilhéus, dezembro de 2010. Embasamento teórico para o manejo populacional do muriqui-do-norte, *Brachyteles hypoxanthus* (KUHL 1820) (PRIMATES, ATELIDAE). Orientador: Daniel Brito.

O gênero *Brachyteles* era considerado um gênero monoespecífico até a metade do século XX, quando foram reconhecidas duas espécies, muriqui-do-norte (*B. hypoxanthus*) e o muriqui-do-sul (*B. arachnoides*). Delas, o muriqui-do-norte se encontra em maior risco de extinção, sendo considerada uma das espécies de primatas mais ameaçadas do planeta. Tratando-se de uma espécie endêmica da Mata Atlântica brasileira e considerando seu crítico status para a conservação, este trabalho buscou contribuir com a conservação da espécie explorando as bases teóricas para o seu manejo. Especificamente, os objetivos foram avaliar os efeitos da dispersão sobre diferentes cenários de subdivisão da população mínima viável da espécie, analisar teoricamente a viabilidade dos seus doze remanescentes populacionais confirmados e avaliar as principais opções de manejo, considerando estimativas de custos de implementação. Para isso, um total de 336 cenários foram elaborados e simulados empregando-se o programa Vortex. As projeções demonstram que a subdivisão da população mínima viável não pode ser compensada por taxas de dispersão, sendo que apenas os cenários com duas subpopulações ligadas a uma baixa taxa de dispersão foi capaz de render uma dinâmica populacional próxima daquela exibida por uma população não subdividida. Os modelos construídos para avaliar a persistência dos remanescentes populacionais indicam que apenas cinco exibem viabilidade em longo prazo (i.e. > 50 gerações); cinco apresentam estabilidade demográfica, mas são projetados elevados riscos genéticos em longo prazo; duas estão sob forte ameaça da instabilidade demográfica e genética; uma é inviável mesmo em curto/médio prazo. Das quatro populações viáveis, duas estão em áreas que em estudos anteriores foram consideradas capazes de manter populações da espécie em longo prazo. As principais opções de manejo sugeridas na literatura para a espécie foram simuladas para os remanescentes que demandam por intervenções (aqueles inviáveis ou instáveis). Os programas de manejo simulados compreenderam uma extensão temporal de 10 anos. Foram identificadas 5 populações com o potencial de doação de indivíduos. Dentre as demais, a incorporação de hábitat foi a que mais frequentemente destacou-se como a melhor opção. Os resultados clarificam os principais aspectos teóricos para o manejo e conservação de *Brachyteles hypoxanthus* e serão importantes em novas discussões envolvendo a espécie.

ABSTRACT

EDUARDO, Anderson Aires, M.S., Universidade Estadual de Santa Cruz, Ilhéus, dezembro de 2010. Theoretical basis for population management of northern muriqui, *Brachyteles hypoxanthus* (KUHL 1820) (PRIMATES, ATELIDAE). Advisor: Daniel Brito.

The genus *Brachyteles* was considered a monospecific until the mid-twentieth century, when two species were recognized, the northern muriqui (*B. hypoxanthus*) and the woolly spider monkey (*B. arachnoides*). Of them, the northern muriqui exhibit greater risk of extinction and is considered one of the most endangered primate on the Earth. Considering its condition of endemic species of the Brazilian Atlantic Forest and its critical conservation status, this study aimed to contribute with the conservation of this species through exploring the theoretical basis for its management. Specifically, the objectives were to evaluate the effects of dispersion on different subdivision scenarios of northern muriqui minimum viable population, to analyze theoretically the viability of the thirteen remaining populations and assess the main management options for the focal species, considering cost estimates for implementation. For this, a total of 336 scenarios were developed and simulated using the software Vortex. The projections show that the subdivision of minimum viable population cannot be offset by dispersal rates, and only the scenarios with two subpopulations and a low rate of dispersal was able to yield a population dynamics close to that exhibited by a not subdivided population. The models developed to evaluate the persistence of remnant populations indicate that only five exhibit long-term viability (i.e. > 50 generations); five were demographic stable, but are projected high genetic risk in long term; two are under severe threat of demographic and genetic instability; one is not viable even in the short to medium term. Of the five viable populations, two are in areas considered able to maintain populations of species in the long term. The main management options suggested in the literature were simulated for the seven remnants that demand for interventions (those are not viable demographically or genetically). Management programs included a simulated temporal extension of 10 years. We identified the five viable populations as potential source of individuals for translocation/reintroduction programs. Among the others, the incorporation of habitat was the most frequently best management option. Taken together, the results clarify the main theoretical aspects for management and conservation of *Brachyteles hypoxanthus* and will be important in further discussions involving the conservation of this species.

INTRODUÇÃO

O gênero *Brachyteles* foi considerado monoespecífico até a metade do século XX, quando Vieira (1944) sugeriu o reconhecimento de duas espécies para o táxon. Esta diferenciação foi ignorada, inclusive pelo próprio autor, em trabalhos subsequentes e apenas mais recentemente foi reconhecida a existência das espécies *Brachyteles hypoxanthus* e *Brachyteles arachnoides (*Assumpção 1983; Coimbra-Filho 1990; 1992a; 1992b; Lemos de Sá & Glander 1993). Estas espécies podem ser diferenciadas morfologicamente através da pigmentação da face, que é negra em *B. arachnoides* e clara em *B. hypoxanthus*, além da ausência de um polegar vestigial nesta última espécie (Aguirre 1971; Strier 1992; Rylands et al. 1995, 2000; Groves 2001). *Brachyteles hypoxanthus* tem sua distribuição geográfica abrangendo os estados de Minas Gerais, Bahia e Espírito Santo, enquanto *B. arachnoides* os estados do Paraná, São Paulo e Rio de Janeiro (Bicca-Marques 2006), o que justifica em parte os nomes populares destas espécies: muriqui-do-norte e muriqui-do-sul, respectivamente.

Os muriquis são as maiores espécies de primatas neotropicais (Rowe 1996), endêmicos da Mata Atlântica (Aguirre 1971). Historicamente, habitavam várias formações fisionômicas deste bioma, como as matas litorâneas da Serra do Mar até florestas semidecíduas interioranas. Com relação ao *status* de conservação, o muriqui-do-sul, *Brachyteles archnoides*, encontra-se na categoria “ameaçado” (IUCN 2010), enquanto o muriqui-do-norte, *Brachyteles hypoxanhtus*, é “criticamente ameaçado” (IUCN, 2010). O muriqui-do-norte é considerado um dos primatas em maior risco de extinção do planeta (Strier et al. 2006), sendo atualmente reconhecidas apenas 13 populações remanescentes distribuídas em fragmentos florestais de Minas Gerais e Espírito Santo (Oliver & Santos 1991; Rylands et al. 2000; Mendes 1994).

Semelhante a maior parte das espécies que apresentam algum risco de extinção, duas das maiores ameaças para persistência dos muriquis-do-norte na natureza são a fragmentação e a perda de hábitat (Mendes et al. 2010). Tais perturbações modificam a dinâmica populacional, tornando-as mais suscetíveis à estocasticidade dos processos demográficos e genéticos (Gilpin & Solé 1986). Uma das principais ferramentas para estudar populações neste contexto é a Análise de Viabilidade Populacional (AVP). Ela compreende uma abordagem estratégica para projetar o destino mais provável das populações que apresentam este tipo de dinâmica e foi empregada neste trabalho para investigar a existência de uma configuração metapopulacional capaz de garantir a persistência da população mínima viável para o muriqui-do-norte (PMV) (Capítulo 1), analisar a viabilidade dos remanescentes populacionais desta espécie (Capítulo 2) e avaliar as principais estratégias de manejo populacional (Capítulo 3).

Estudos que empregam a AVP tornaram-se muito utilizados nas últimas décadas, o que desencadeou uma série de discussões sobre a acurácia desta ferramenta de modelagem populacional. Basicamente, enquanto alguns trabalhos afirmam a utilidade destes modelos para realização de predições quantitativas sobre a persistência de populações (ver Brook et al. 2000 e Book et al. 2002), outros argumentam que tais predições devem ser vistas com muita cautela e que sua aplicabilidade é bastante limitada (ver Ludwing 1999; Fieberg & Ellner, 2000; Coulson et al. 2001 e Ellner et al. 2002). Contudo, ambas as partes concordam quanto à aplicabilidade das AVP's na avaliação de opções de manejo para populações (ver Coulson et al. 2001). Neste sentido, *13* cenários foram elaborados e comparados no Capítulo 1, os quais consistiram em aplicar diferentes combinações de subdivisão e dispersão aos tamanhos mínimos para a viabilidade demográfica e genética da espécie focal. No Capítulo 2, *223* cenários foram criados para o desenvolvimento da análise de viabilidade das 13 populações de *Brachyteles hypoxanthus*. No Capítulo 3, *68* cenários foram desenvolvidos para que as principais linhas de ação para o manejo populacional da espécie (manejo do hábitat e suplementação de indivíduos) fossem avaliadas, considerando seus benefícios e custos. Nesta dissertação, cada um destes capítulos é apresentado em língua inglesa, editados dentro dos moldes dos periódicos para os quais cada um foi submetido separadamente (antes da data da defesa).

Em seu conjunto, os resultados gerados pelas simulações permitirão que a tomada de decisão para o manejo do muriqui-do-norte seja otimizada, guiando o melhor caminho para o investimento dos parcos recursos destinados a conservação da biodiversidade.

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CAPÍTULO 1

MODELLING THE IMPACTS OF POPULATION SUBDIVISION ON THE VIABILITY OF *Brachyteles hypoxanthus*

(Submetido ao periódico Biodiversity and Conservation em 30 de dezembro de 2010)

Modelling the impacts of population subdivision on the viability of *Brachyteles hypoxanthus*

Eduardo A. A.[1, 3]

Brito D.[2]

[1] Programa de Pós-graduação em Ecologia e Conservação da Biodiversidade. Departamento de Ciências Biológicas, Universidade Estadual de Santa Cruz, Ilhéus, BA, Brasil.

[2] Departamento de Ecologia. Universidade Federal de Goiás, Goiânia, Brasil.

[3]Corresponding author: andereduardo@yahoo.com.br

Abstract - Among the anthropogenic contributions to the contemporary biodiversity crisis, fragmentation and habitat loss pose the most severe. Both processes act synergistically, resulting in the subdivision of populations experiencing conditions of continuous habitat, converting them into small clusters of individuals in patches of habitat. The dynamics of these small populations differ intrinsically from those more abundant, being that demographic, environmental and genetic stochastic factors constituting a serious risk. This work use Population Viability Analysis to explore the potential effects that different subdivision and dispersal scenarios have in the demographically (MVPd) and genetically (MVPg) minimum viable population for persistence of neotropical primate *Brachyteles hypoxanthus*. The results demonstrate that subdivisions and dispersal rates interact in the definition of a threshold for metapopulation collapse which is easily achieved as smaller are subpopulations and larger are the dispersal rates. Only the scenarios with two subpopulations and small dispersal rates yield a population dynamics close to the baseline scenario (not subdivided MVPs). Other scenarios exhibit a negative effect on the metapopulation persistence, mainly on the population growth rate and its variation along the time. Finally, metapopulation configurations do not generate benefits for the MVPs and should be employed only in cases where the maintenance of a unique and viable population is impractical.

Key words: Atlantic Forest; fragmentation; habitat loss; metapopulation dynamics; Population Viability Analysis; VORTEX.

Introduction

Habitat loss and fragmentation are the main threats to the world's biodiversity (Wilcox and Murphy 1985; Fahrig and Merriam 1994; Bowers et al. 1996; Fahrig 1997, 2001; Gaines et al. 1997). One of the consequences of habitat fragmentation is population subdivision, where one (or a few) large population inhabiting continuous habitat is converted into several small populations (or sub-populations) inhabiting the remnants of native habitat (Burkey 1995; Fahrig and Merriam 1994; Fahrig 1997, 2001). It is already known that small populations are more prone to suffer from stochastic events and processes (Gilpin and Soulé 1986), and because of that, may decline towards extinction (Gilpin and Solé 1986; Hanski et al. 1996). The study of such extinction vortex is a cornerstone of conservation biology (Gilpin and Solé 1986; Sheffer 1981; Soulé 1987).

The Atlantic Forest harbors a significant amount of endemic species and it is one of the world's ecosystems that suffered most with habitat loss and fragmentation, being listed as a Biodiversity Hotspot (Mittermeier et al. 2005). Nowadays, estimates show that only between 11% and 16% of the original native vegetation cover is left (Ribeiro et al. 2009). Brazil's major economic centers are found within the biome borders (Jacobsen 2003; Pinto and Brito 2003; Young 2003). Besides that, most of the country's population (approximately 169 million, or 70% of the country's population), lives in the Atlantic Forest domain (Jacobsen 2003; Pinto and Brito 2003; Young 2003). As a result, what is left of the Atlantic Forest is scattered as small and isolated forest remnants (Ribeiro et al. 2009).

Even though it has already been demonstrated that the maintenance of large remnants within a fragmented landscape is important for preserving community structure (Chiarello 1999; 2000a, b), we are just beginning to comprehend how changes in population structure, as a result of subdivision, might affect population persistence in fragmented landscapes (e.g. Lindenmayer et al. 1993; Lacy and Lindemayer 1995; Lindemayer and Lacy 1995a, b; Brito and Fonseca 2007). In this sense, only in the last decades we became able to understand the role of small habitat remnants for maintaining native populations within a dynamic landscape (Turner and Corlett 1996; Chiarello 2000b). Considering the demand for the conservation of the Critically Endangered northern muriqui (*Brachyteles hypoxanthus*) (IUCN 2010) in the fragmented Atlantic Forest landscapes, in this study our objectives was to model the impacts of subdivision on the viability, demography and genetic structure of the minimal viable population of such species.

Methods

Target species

The muriquis (genus *Brachyteles*) are the largest Neotropical primates (Aguirre 1971; Rowe 1996). *Brachyteles hypoxanthus* is endemic to the Atlantic Forest (Aguirre 1971). It has an herbivorous diet (Milton 1984; Carvalho et al. 2004; Talebi et al. 2005), but it also consumes seeds, pollen and bamboo (Strier 1991a). Northern muriquis are diurnal and arboreal, but they may descend to the ground to cross gaps, drink water and sometimes, to play (Dib et al. 1997). They are social and groups may have up to 50 individuals (Strier 1993, 1994), with males showing a philopatric behavior and females dispersing when reaching sexual maturity (six years old) (Strier and Ziegler 2000; Brito and Grelle 2006). Mean age at first reproduction is estimated as seven years for males and nine years for females (Strier and Ziegler 2000; Strier 2005). Litter size is one and there is a three year inter-birth interval (Strier 1996). Longevity is estimated as 35 years (Rylands et al. 1998).

PVA model and scenarios

To model our scenarios, we used the software VORTEX (version 9.92), a Monte Carlo simulation of the effects of deterministic forces as well as demographic, environmental and genetic stochasticity and catastrophes on the dynamics of wildlife populations (Lacy 1993, 2000; Lacy et al. 2008).

Brito and Grelle (2006) identified critical population sizes for the persistence of *Brachyteles hypoxanthus*. A population of 40 individuals should ensure safety from demographic stochasticity (MVPd) and a population of 700 individuals should ensure safety against genetic stochasticity (MVPg) (Brito and Grelle 2006). Given these data, the dynamics of populations of *B. hypoxanthus* with 40 and 700 individuals, and metapopulations where the original population size (40 and 700) was subdivided into 1, 2, 4, and 8 (and 16, for the MVPg) populations, were explored. The dispersal rate among populations was modeled in terms of individual probability. Thus, four scenarios with the individual probabilities of dispersal of 0, 0.05, 0.10 or 0.20 per year were simulated, giving a total of 30 scenarios. We completed 500 iterations for each scenario. Time scale is of paramount importance in extinction studies (Armbruster et al. 1999; Frankham and Brook 2004) and the time frame of 50 generations was selected according to the biology of the target species (Armbruster et al. 1999). 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; Brito et al. 2008) (see appendix). A population is considered demographically viable if it presents <10% extinction probability in 50 generations; and genetically viable if it retains >90% of its original genetic diversity in 50 generations.

Results

The general results follow the same trends for subdivision scenarios both for the large and small population sizes analyzed (Tables 1 and 2). As expected, no extinctions were observed when we modeled single continuous population of 40 or 700 individuals (Table 1). However, when we modeled scenarios including population subdivision, population growth rate and final metapopulation size decreased (Tables 1 and 2). On the other hand, population fluctuation and probability of extinction increased (Table 1). The results show that dispersal rates have limited power to counter the negative effects of population subdivision. The increased dispersal rates only showed positive effects on population viability in the scenario with the smallest rate of subdivision (2 populations) for the subdivision of a population of 40 individuals (Table 1), and when subdivision was more severe, the increase in dispersal rates was not capable to compensate deleterious effects from fragmentation (Table 1).

Our results show that the only viable subdivision scenario for a small population of northern muriquis (40 individuals) is the one with two subpopulations (Table 1). The scenarios evaluating the subdivision of a large population (700 individuals), also shows that subdividing the original population into two subpopulations do not affect long-term persistence, regardless of the dispersal rates among them (Table 2). However, as subdivision intensifies, scenarios with higher dispersal rates become not viable (Table 2). Dispersal-generated metapopulation stochasticity reaches its maximum negative effect in scenarios where large populations (700 individuals) is subdivided into 16 sub-

Table 1: Results of the subdivision simulations for the northern muriqui MVPd (40 animals) in VORTEX.

Scenarios	**Dispersal rate** [1]	**Population growth rate (*r*) [mean(SD)]**	**Probability of extinction (PE) [mean(SD)]**	**Final population size (N) [mean(SD)]**	**Heterozygozity (He) [mean(SD)]**	**Time to extinction** [2]
1 x 40	-	0.061 (0.067)	0.000	39.33 (2.01)	0.178 (0.209)	-
2 x 20	0.00	0.059 (0.065)	0.042	31.74 (10.95)	0.377 (0.227)	678.4
	0.05	0.051 (0.064)	0.008	34.39 (8.02)	0.192 (0.213)	658.5
	0.10	0.045 (0.064)	0.006	35.47 (6.78)	0.219 (0.217)	695.7
	0.20	0.031 (0.065)	0.032	35.14 (7.65)	0.254 (0.219)	737.3
4 x 10	0.00	0.046 (0.088)	0.980	0.20 (1.36)	0.000 (0.000)	369
	0.05	0.019 (0.083)	0.954	0.59 (2.94)	0.050 (0.136)	451.4
	0.10	0.003 (0.085)	1.000	0.00 (0.00)	0.000 (0.000)	226.5
	0.20	-0.018 (0.093)	1.000	0.00 (0.00)	0.000 (0.000)	84.2
8 x 5	0.00	0.017 (0.096)	1.000	0.00 (0.00)	0.000 (0.000)	100.4
	0.05	-0.015 (0.100)	1.000	0.00 (0.00)	0.000 (0.000)	83.7
	0.10	-0.026 (0.101)	1.000	0.00 (0.00)	0.000 (0.000)	56.5
	0.20	-0.025 (0.098)	1.000	0.00 (0.00)	0.000 (0.000)	40.2

[1] The dispersal rate is measured in individual probability of dispersal per year.

[2] Time to extinction is measured in years.

Table 2: Results of the subdivision simulations for the northern muriqui MVPg (700 animals) in VORTEX.

Scenarios	**Dispersal rate** [1]	**Population growth rate (r) [mean(SD)]**	**Probability of extinction (PE) [mean(SD)]**	**Final population size (N) [mean(SD)]**	**Heterozygozity (He) [mean(SD)]**	**Time to extinction** [2]
1 x 700	-	0.062 (0.050)	0.000 (0.000)	698.63 (7.99)	0.910 (0.019)	-
2 x 350	0.00	0.063 (0.041)	0.000 (0.000)	696.39 (9.52)	0.914 (0.018)	-
	0.05	0.056 (0.041)	0.000 (0.000)	696.88 (9.27)	0.916 (0.018)	-
	0.10	0.049 (0.041)	0.000 (0.000)	696.27 (9.84)	0.921 (0.018)	-
	0.20	0.034 (0.039)	0.000 (0.000)	691.04 (13.41)	0.929 (0.014)	-
4 x 175	0.00	0.063 (0.035)	0.000 (0.000)	696.45 (9.22)	0.922 (0.013)	-
	0.05	0.042 (0.034)	0.000 (0.000)	690.43 (11.52)	0.927 (0.014)	-
	0.10	0.020 (0.032)	0.000 (0.000)	679.21 (17.87)	0.932 (0.014)	-
	0.20	-0.023 (0.063)	1.000 (0.000)	0.00 (0.00)	0.000 (0.000)	207.2
8 x 87	0.00	0.063 (0.032)	0.000 (0.000)	694.36 (9.21)	0.934 (0.008)	-
	0.05	0.013 (0.029)	0.000 (0.000)	652.09 (24.07)	0.932 (0.014)	-
	0.10	-0.034 (0.063)	1.000 (0.000)	0.00 (0.00)	0.000 (0.000)	137.8
	0.20	-0.050 (0.067)	1.000 (0.000)	0.00 (0.00)	0.000 (0.000)	77.5
16 x 44	0.00	0.063 (0.030)	0.000 (0.000)	693.6 (10.17)	0.951 (0.003)	-
	0.05	-0.039 (0.062)	1.000 (0.000)	0.00 (0.00)	0.000 (0.000)	116.5
	0.10	-0.05 (0.067)	1.000 (0.000)	0.00 (0.00)	0.000 (0.000)	76.3
	0.20	-0.049 (0.067)	1.000 (0.000)	0.00 (0.00)	0.000 (0.000)	77.0

[1] The dispersal rate is measured in individual probability of dispersal per year.

[2] Time to extinction is measured in years.

populations, where metapopulation is viable only if isolated (Table 2).

Discussion

This study sought to explore the effects of population subdivision on *Brachyteles hypoxanthus*. Theoretical studies indicate that population subdivision negatively affects demographic and genetic parameters of wildlife populations (Shaffer 1981). PVA models based on empirical data also showed such trend for some mammals (e.g. Lacy and Lindenmayer 1995; Lindenmayer and Lacy 1995a; Brito and Fonseca 2007; Brito 2009a), and our results corroborate these previous observations. In a metapopulation structure, the size of the constituent populations seems to be an important factor for metapopulation long-term persistence (Fahrig 1997, 1998, 2001; Hanski and Gaggiotti 2004). Regarding dispersal rates, our results clearly show a threshold above which dispersal collapses the metapopulation structure. Until such point, only the heterozygosity was affected positively. Thus, below such threshold, we can understand the dispersal rate as partially compensating subdivisions once affect negatively the demography but favors the genetic of the metapopulation.

Demographic stability was not affected by the levels of population subdivision modeled for the large population (700 individuals), even when there was no dispersal and subdivided populations were isolated. This could be explained by the fact that in these scenarios, each subpopulation size resulting from the subdivision process is larger than the minimum size required for demographic stability in this species (Brito and Grelle 2006). However, the subdivision of small populations (40 individuals) was detrimental to their long-term persistence, depressing growth rate and increasing demographic stochasticity (Shaffer 1981; Soulé 1987; Brito 2009b), and enhancing connectivity through dispersal had the counter-intuitive result of further increasing population instability through metapopulation stochasticity (Lindenmayer and Lacy 1995a; Brito and Fonseca 2007; Brito 2009a).

Generally, it is accepted that dispersal of individuals among populations has a positive effect on persistence (Grant and Grant 1992; Simberloff et al. 1992; Lande 1998; Frankham 1995; Frankham 1999; Waite et al. 2005), a process known as the rescue-effect (Brown and Kodric-Brown 1977). However, there is evidence that for small mammals, a threshold exists, below which stochastic events through metapopulation instability have a more powerful grip on population dynamics and dispersal might be detrimental to persistence (Lindenmayer and Lacy 1995a, 1995c; Lacy and Lindenmayer 1995; Brito and Fonseca 2007; Brito 2009a). Most metapopulation studies on mammals focus on small species (Lambin et al. 2004), and there are relatively few metapopulation studies on primates (e.g. Swart and Lawes 1996; Cowlishaw and Dunbar 2000; Chapman et al. 2003; Mandujano and Escobedo-Morales 2008). Mandujano and Escobedo-Morales (2008), studied populations of *Alouatta palliata mexicana* in a fragmented landscape and did not observe negative effects of dispersal on population persistence. However, we observed such detrimental effects for *Brachyteles hypoxanthus*, suggesting this process may also affect medium and large mammals, and we speculate that possibly the *Alouatta palliata mexicana* case (Mandujano and Escobedo-Morales 2008) took place within a landscape that was still above the threshold for the species, a scenario when theory shows dispersal is expected to have positive or no effects on persistence.

The Atlantic Forest is one of the ecosystems most impacted by habitat loss and fragmentation in the world (Myers et al. 2000; Mittermeier et al. 2005). The investigation of how landscape modification might affect population structure and dynamics, and consequently species persistence in such modified landscapes is of paramount importance.

Our results corroborate previous theoretical and empirical evidence that high dispersal rates may be detrimental to populations due to stochastic events taking place in highly fragmented landscapes. These analyses warn that caution must be taken when devising conservation actions in such scenarios, as commonplace strategies to counter fragmentation to this date are based on improving connectivity through dispersal corridors.

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Appendix

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 · age · 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 · age · 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 R0 = 3.402

Generation time for: females = 19.57 males = 17.87

CAPÍTULO 2

USING VIABILITY MODELS TO GUIDE POPULATION MANAGEMENT FOR THE NORTHERN MURIQUI (*Brachyteles hypoxanthus*)

(Submetido ao periódico Oryx em 30 de dezembro de 2010)

Using viability models to guide population management for the northern muriqui (*Brachyteles hypoxanthus*)

Eduardo A. A.[1, 3]

Brito D.[2]

[1] Programa de Pós-graduação em Ecologia e Conservação da Biodiversidade. Departamento de Ciências Biológicas, Universidade Estadual de Santa Cruz, Ilhéus, BA, Brasil.

[2] Departamento de Ecologia. Universidade Federal de Goiás, Goiânia, Brasil.

[3]Corresponding author: andereduardo@yahoo.com.br

Abstract: Population ecology has contributed greatly to the efforts towards the management and conservation of biological diversity. In particular, Population Viability Analysis (PVA) has provided important insights to the study of population dynamics and extinction processes. Considering the demand for theoretical contributions that allow the optimization of decision making in Brazilian conservation, this study evaluated the viability of remaining populations of northern muriqui, *Brachyteles hypoxanthus*. VORTEX was used for population modeling. The results show that only five of the thirteen populations confirmed in nature are viable in long-term (i.e. 50 generations), five are demographically stable but are projected to suffer high genetic risk in the long term, two are under severe genetic and demographic threat. The incorporation of habitat was the priority management strategy for the majority of populations that are in demand for interventions. Translocation and reintroduction programs are important research strategies to repopulate areas where the species was extinct but viable populations could be maintained. Expeditions for discovering new populations also represent strategic actions for the conservation of species and should be implemented in addition to management options proposed in this study for the remaining populations.

Keywords: Vortex, AVP, conservation, extinction, Atlantic Forest, translocation, reintroduction, primates.

Introduction

There are innumerable causes that determine the extinction of a species in nature, and the increase in human activities has created a series of new anthropogenic-driven processes that may drive species to extinction (Lawton, 1994). Given such complexity, theoretical approaches can be very useful for guiding intellectual and economic investments to conservation. Population Viability Analysis (PVA) has provided important insights to the study of population dynamics (Lande, 2002), becoming a valuable tool for conservation of biodiversity (Lindenmayer et al., 1995; Brook, 2000; Brook et al., 2000; Brook et al., 2002). Despite the extensive debate about its limitations, it is common sense the benefits of its application in studies that aim to compare populations, especially in response to different management strategies (Coulson et al., 2001; Brook et al., 2002). Its use has increased over the years and it is recurrently employed to help decision making in the management of the scarce conservation resources (Brook et al., 1999; Brook et al., 2000).

In Brazil, only recently PVA has been used both in theoretical and applied conservation research (Brito, 2009). Just four theoretical studies were published, and of the 767 Brazilian species threatened by extinction (IUCN, 2010), only ten were covered with this type of analysis (Brito, 2009). Considering the current environmental and socio-economic situation of this mega-diverse country, with two biodiversity hotspots (Atlantic Forest and Cerrado) (Mittermeier et al., 2005) and three wilderness areas (Amzônia, Caatinga and Pantanal) (Mittermeier et al., 1998, 2002), the development of further studies on population viability can yield important contributions to the management of biological diversity in Brazil.

Among Brazilian biodiversity, primates stand out as one of the *taxa* most threatened by the current extinction wave (Lyles & Dobson, 1989; Mittermeier et al., 2007; IUCN, 2010). Besides the demand for its conservation, the vulnerability of this group has motivated interest in its use in studies of extinction processes and conservation (Dobson & Lyles, 1989; Gibbons & Harcourt, 2009). In this context, we aimed in this work to analyze the viability of the remaining populations of northern muriqui, one of the most threatened primates in the planet (Strier et al., 2006), and explore potential options for managing them.

Matherials and Methods

Aspects of Brachyteles hypoxanthus Natural History

The muriquis (genus *Brachyteles*) are the largest Neotropical primates (Aguirre, 1971). Individuals of the focal species (*Brachyteles hypoxanthus*) have around 60 cm in length (excluding tail) and weight about 12 kg (Rowe, 1996). Muriquis are endemic to the Atlantic Forest and historically they inhabited a wide variety of vegetation types, from the coastal forests of the Serra do Mar to the semi-deciduous forests inland (Aguirre, 1971). It is considered an herbivorous species (Milton, 1984, Carvalho et al., 2004; Talebi et al., 2005) and may also consume seeds, pollen, bamboo and ferns (Strier, 1991a). The northern muriqui is diurnal and arboreal, but may go to the ground to cross gaps, drink water and play (Dib et al., 1997). They can live in groups of more than 50 individuals (Strier, 1993-1994), whose males are philopatric and females disperse when they are 6 years old (Strier & Ziegler, 2000). The first offspring occurs at 7 years of age for males and 9 for females (Strier & Ziegler, 2000; Strier, 2005). Only one offspring is generated per brood, with a mean period of approximately three years between broods (Strier, 1996). At 35 years old, on average, muriquis reach their maximum reproductive age (Rylands et al., 1998). The mean density for this species is 0.0605 (± 0.0899) individuals per hectare (Strier & Fonseca, 1996/1997).

PVA

To perform the analysis, we used the software VORTEX (version 9.92) (Lacy et al., 2008). This software is one of the most used modeling tools in population viability analysis (Lindenmayer et al., 1995). VORTEX uses a Monte Carlo process for simulating the effects of deterministic and stochastic forces on the dynamics of populations (Lacy, 1993; Lacy, 2000). The iterations reproduce a typical sequence of events in the life cycle of organisms, such as selection of pairs for breeding, reproduction, mortality, aging, migration, , among others (Lacy, 1993 ; Lacy, 2000). Lacy (1993; 2000) present a detailed description of the software.

Simulations and scenarios

According to Mendes et al. (2005), there are 13 extant populations of *Brachyteles hypoxanthus* confirmed in nature (Table 1). The input data for the model were compiled from previous population studies available in the literature (Strier, 1991b, 1993/1994, 2000, 2005, 2006, Rylands et al., 1998; Brito & Grelle, 2006; Coutinho, 2007; Brito et al., 2008). The output parameters used in the analysis were the probability of extinction (*PE*), heterozygosity (*He*) and the rate of population increase (*r*). *PE* is defined as the proportion of extinct populations across the iterations, and *He* is the proportion of the initial genetic diversity that remains at the end of simulations. It was assumed that a population is demographically viable when probability of extinction was equal to or less than 1%. A population was considered genetically viable when it maintained at least 90% of its original heterozygosity (Shaffer, 1981). Thus, populations with $PE = 1$ and $He = 0$ were classified as unviable. Those that exhibited $0.01<PE<1$ and $0<He<0.90$

were classified as demographically and genetically unstable. If $PE<0.01$ and $0<He<0.9$, the population was considered to be genetically vulnerable, and if $PE<0.01$ and $He>0.90$, the population was considered viable.

Tabela 1: Confirmed populations of northern muriqui (data from Mendes et al., 2005 and Cunha et al., 2009).

Locality[1]	**Area (ha)**	**Minimum population confirmed**
Alto Cariri National Park (AC)	18000	7
Mata Escura Biological Reserve (MERB)	50890	28
Córrego de Areia Farm (CAF)	494	13
Rio Doce State Park (RDSP)	35976	124
Caratinga Biological Estation (CBS)	957	226
Augusto Ruschi Biological Reserve and adjacent forest remnants (ARBR)	3573	14
Santa Maria de Jetibá fragments (SMJ)	2000[2]	84
Esmeralda farm (EF)	44	3
RPPN Mata do Sossego (MS)	180+620[3]	41
Caparaó National Park (CNP)	31853	82
Serra do Brigadeiro State Park (SBSP)	13210	226
Ibitipoca State Park (ISP)	1488[4]	7
Itatiaia Nacional Park (INP)	28155	16

[1] In parentheses are the abbreviations that will be used throughout the text.
[2] Area constituted for a group of 13 forest fragments.
[3] The area registered for the RPPN Mata Sossego is 180 ha. However, with an adjacent forest fragment (620 ha), form a single block of 800 ha.
[4] The predominant vegetation in the area is the high altitude grassland and only about 25% of the area is covered by forest.

To investigate the robustness of the emerging scenarios, a sensitivity analysis was performed through modifications in different model parameters. This procedure allows diagnosing what are the most important parameters of the model, assisting in the interpretation of the results against the uncertainty in parameter estimation and the differences inherent in different populations (McCarthy et al., 1995). Thus, the parameters used were: (1) mortality, (2) sex ratio, (3) availability of reproductive females, (4) inbreeding depression, (5) density (simulated as different carrying capacities - *K*), (6) initial number of individuals in the population, and (7) age of females at the beginning of reproductive life. To SMJ, where the population is distributed in small isolated forest patches, an additional scenario was simulated where dispersion rates of 0 and 5% were tested.

For the mortality rate we simulated scenarios with 5, 10 and 20% of increase over the baseline. For the sex ratio were simulated scenarios with 0.5 and 0.65 males per female (Rylands et al., 1998). The model sensitivity to the availability of reproductive females was analyzed by scenarios with values of 20% and 33% (Rylands et al., 1998). For inbreeding depression, a scenario was created including the occurrence of this process. The sensitivity to density was measured using 10, 20 and 40% of increase in carrying capacity. For the initial population, the scenarios were created with 10, 20 and 40% of individuals, and finally the beginning of reproductive age has been increased and reduced by one year to examine the model sensitivity to this input data. The significance of such scenarios was tested by *t* tests (Zar, 1996).

Results

Viability of populations

The probability of extinction was estimated at zero or near zero for ten populations (MERB, CAF, RDSP, ECB, ARBR, SMJ, CNP, INP, SBSP, MS). Of the remaining three, AC exhibited *PE* slightly over 1%, ISP has approximately 10% and for EF it was 100%. Conversely, these generalized low values of *PE* do not reflect the great genetic risk faced by the populations, and only five were capable of holding at least 90% of its initial heterozygosity at the end of simulations (MERB, RDSP, CNP, INP, SBSP). The population growth rates did not vary significantly among populations.

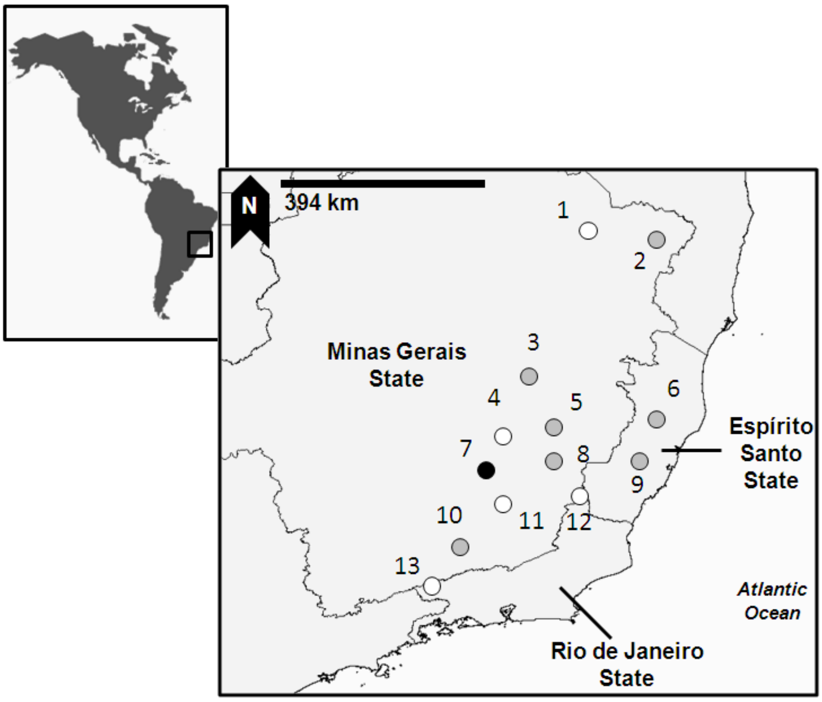

Figure 1: Results of Viability Analysis in VORTEX for the 13 *Brachyteles hypoxanthus* populations. Black circles = unviable population; Gray circle = demographically and/or genetically unstable; White circle = viable populations. 1 = MERB; 2 = AC; 3 = CAF; 4 = RDSP; 5 = CBS; 6 = ARBR; 7 = EF; 8 = MS; 9 = SMJ; 10 = ISP; 11 = SBSP; 12 = CNP; 13 = INP

Sensitivity analysis

Alto Cariri National Park (AC)

The *PE* was affected by all the simulated scenarios, and the only exception was the 5% increase in mortality rate. The scenarios for sex ratio and availability of females were capable to affect *r* and *He*. Improvements in carrying capacity and in the number of animals in the population also generate important impacts in *He*. Thus, the management options that aim to combine increase the carrying capacity and the number of individuals in the population will be able to generate the best positive effects on the whole population dynamics. Strategies involving the translocation of individuals, and feeding supplementation are the main options for the management of this population.

Mata Escura Biological Reserve (MERB)

The population of MERB was demographically (*PE*) and genetically (*He*) viable in all scenarios analyzed. Considering the robustness of this population to increased mortality rate, one can envision its potential as a source of individuals for translocation and establishment of captive populations.

Córrego de Areia Farm (CAF)

PE, *He* and *r* were sensible to the scenarios for sex ratio and availability of reproductive females. *He* and *PE* were also sensitive to the scenarios for changes in the carrying capacity. As the scenarios for carrying capacity improvement affect positively *He* and *PE* while keeps *r* stable, strategies aimed at increasing the quality of habitat (e.g. food supplementation and reforestation) should be targets for the management of this population.

Rio Doce State Park (RDSP)

Both the demographic (*PE*) and genetic (*He*) viability were insensitive to changes in the model. Similar to the results for the population of the MERB, its robustness to population mortality rates makes it a potential source of individuals for translocation programs or implementation of captive populations.

Caratinga Biological Station (CBS)

The *PE* was insensitive to all the scenarios. However, *r* was sensitive to sex ratio, inbreeding depression and the availability of females. *He* was sensible to any parameter tested and should be interpreted as a critical aspect for population management. As reforestation is the most economically and logistically feasible intervention and considering that carrying capacity improvement have positive influence in *He* (keeping stable *r* and *PE*), management options involving the improvement of habitat quality (e.g. food supplementation and reforestation) can be the best strategy for managing this population.

Augusto Ruschi Biological Reserve (ARBR)

The demographic viability was affected only by the sex ratio scenario. With the exception of inbreeding depression and the mortality rate, *He* was sensitive to all other scenarios. Sex ratio, inbreeding depression and the availability of females were also able to affect *r*. As increasing the carrying capacity of the population keeps *r* and *PE* stable but positively affect *He*, strategies involving the increase of habitat quality for the species should be prioritized.

Santa Maria de Jetibá (SMJ)

PE was sensitive to all scenarios, and the sex ratio scenario was also to affected *He* and *r*. All scenarios of dispersal between fragments affected metapopulation dynamics negatively. However, the fusion of the fragments closest associated with rates of 5% dispersion yielded the best results. The union of the nearest populations should be considered the first step in the management of this population.

Esmeralda Farm (EF)

The results indicate that the Esmeralda Farm is unable to maintain a viable muriqui population, even for short periods of time. Even scenarios with increased capacity and availability of females yielded values of *PE* and *He* equal zero. The removal of these individuals for translocation or reintroduction programs is the best management option for this population.

Caparaó National Park (CNP)

Both genetic and population viability were insensitive to the scenarios, but *r* was affected by the sex ratio and the availability of reproductive females. This population can be considered as a potential source of individuals for management programs involving translocation and reintroduction.

Serra do Brigadeiro State Park (SBSP)

Similar to the CNP, the population in SBSP also remained viable over the sensitivity analysis scenarios. The population growth rate (*r*) was sensitive to scenarios for availability of females and sex ratio. This population may also be considered as a potential source of individuals for translocation and reintroduction.

Ibitipoca State Park (ISP)

PE was sensitive to all scenarios. However, only sex ratio affected both *He* and *r*, allowing a glimpse its critical importance for population dynamics. The scenario for the carrying capacity did not affect *r*, but produced positive demographic and genetic effects. As to manipulate animals is a more delicate intervention (and so less desirable) than manipulating habitat, strategies that involve improving carrying capacity should be at the forefront of discussions for the management of this population (e.g. supplemental feeding and reforestation).

Mata do Sossego Private Reserve of Natural Heritage (MS)

PE was stable and viable over the scenarios, but *He* and *r* were susceptible to sex ratio, inbreeding depression and the availability of females. Carrying capacity did not affect *r* and *PE*, but *He* was positively influenced. Considering that MS is demographically stable but has genetic problems, the best strategy for the management of this population should to combine habitat management to improve carrying capacity and supplementation of animals.

Itatiaia National Park (INP)

Changes in parameters did not affect the outputs for the INP population. This population could be considered as a potential source of individuals for management programs involving translocation and reintroduction.

Discussion

It has been previously demonstrated that only five out of 42 protected areas would be able to hold viable populations of *B. hypoxanthus* within the Atlantic Forest (Brito et al., 2008). Of these, only two (CNP and RDSP) maintain extant populations and our results are favorable to the persistence of these populations. Such extant viable populations occupy a prominent position for the conservation of the species. The populations of SBSP, MERB and INP were also considered viable in this work, but the former is not listed as a viable area by Brito et al. (2008) and MERB and INP were not evaluated by these authors. The reason for such discrepancy for SBSP is the difference in the model construction. While in the present study we consider extinction as no individuals in the population, Brito et al. (2008) were more conservative, considering a quasi-extinction threshold of 700 individuals. Regarding MERB and INP populations, their large sizes (about 51000 hectares and 28155 hectares, respectively) suggest that such areas are able to maintain viable populations of northern muriqui, since the minimum area of suitable habitat for the long term viability of these species was estimated around 11500 hectares (Brito & Grelle, 2006). Therefore, these five areas should be considered the main areas for the northern muriqui persistence.

Carrying capacity improvement was the priority intervention for six of the eight populations that require management. These results corroborate Mendes et al. (2005) who put the maintenance and incorporation of habitat areas as a major strategy for the conservation of *B. hypoxanthus*. Initial efforts indicate that this is a feasible option for the management programs. According to Pontual & Boubli (2005), the Muriqui Conservation Project in CBS was able to reforest 30 hectares of forests in six months, with the engagement of local landowners.

The introduction of females appears in sensitivity analysis as an important intervention for the AC. Given the unviability of EF, the individuals in this population could be translocated to AC, in order to improve the dynamics of such population. The population of RDSP, located in the same region, could also serve as a source of individuals for translocation. Regarding AC, individuals could be from MERB, what is located in their vicinity. Besides, Brito & Grelle (2006) suggested CBS as a potential source of individuals for translocation and captivity management, since it is growing (Strier, 2005). The sensitivity analysis indicates that CBS could support the removal of individuals (scenarios for increases in mortality) and be considered a potential source of individuals for management programs involving translocation and reintroduction. In the scientific literature, several authors cite such strategy as an important option for the conservation of muriqui populations (Fagundes, 2005; Mendes et al., 2005; Spot & Boubli, 2005; Brito & Grelle, 2006, Brito et al., 2008). Although empirical studies for the development of specific methodologies are still few, pioneering initiatives have been successful and demonstrate its great potential to be used in future (Mendes et al., 2005).

Four areas listed in Brito et al. (2008) can support northern muriqui viable populations but are currently uninhabited (Una Biological Reserve, Córrego Soberbo e Retiro Special Protection Area, Cariva / Trancoso Environmental Protection Area and Rio Todos os Santos Special Protection Area). The implementation of a captive population program could facilitate the reintroduction of these populations, representing an important step in improving the conservation status of the species (Pissiantti et al., 1998; Pissinatti, 2005). The first experiences with captive breeding of *Brachyteles hypoxanthus* were successful (Coimbra-Filho et al., 1993; Pissinatti, 2005), but new research efforts are still needed before the implementation of a robust management program involving such strategy (Mendes et al., 2005).

Over the past decades, censuses doubled the number of *B. hypoxanthus* populations confirmed in nature (Mendes et al., 2005; Melo & Dias, 2005). Alongside the options examined individually for the 12 populations, the continuation of such studies should be of great importance for the development of management strategies, yielding new population records and allowing that known populations be better studied (Melo & Dias, 2005).

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CAPÍTULO 3

COUPLING VIABILITY AND COST TO GUIDE POPULATION MANAGEMENT FOR THE NORTHERN MURIQUI (*Brachyteles hypoxanthus*)

(Submetido ao periódico Ecological Economics em 30 de dezembro de 2010)

Coupling viability and cost to guide population management for the northern muriqui (*Brachyteles hypoxanthus*)

Eduardo A. A.[1, 3]

Brito D.[2]

[1] Programa de Pós-graduação em Ecologia e Conservação da Biodiversidade. Departamento de Ciências Biológicas, Universidade Estadual de Santa Cruz, Ilhéus, BA, Brasil.

[2] Departamento de Ecologia. Universidade Federal de Goiás, Goiânia, Brasil.

[3]Corresponding author: andereduardo@yahoo.com.br

Abstract: Studies targeting the current biodiversity crisis have generated important theoretical and practical advances in conservation biology. An important emerging aspect is the incorporation of the economic dimension in studies focusing on management practices to conserve biodiversity. In this context, the objective of this study is to evaluate the efficiency and costs of different management options for the northern muriqui (*Brachyteles hypoxanthus*), one of the world's most threatened primates. The viability of thirteen extant populations of the species was estimated using the software VORTEX. We modeled the effects of different management scenarios: (1) addition of habitat at 2% per year, (2) addition of habitat at 5% per year, (3) translocation of 1 female per year, (4) translocation of 5 females per year, (5) supplement 1 female per year from captive populations, (6) supplementation of 5 females per year from captive populations, (7) translocation of 1 female per year + habitat incorporation at 5% per year, (8) translocation 1 female per year + habitat incorporating at 2% per year; (9) translocation of 5 females per year + habitat incorporating at 2% per year. Six populations were identified as potential donors of individuals (Mata Escura Biological Reserve, Rio Doce State Park, Caparaó National Park, Serra do Brigadeiro State Park, Itatiaia National Park and Esmeralda Farm - the first four ones because of long term population viability, the latter for the opposite reason). The scenarios for habitat management produced the better benefits and costs for five of the seven populations that demand for management interventions (Mata do Sossego Particular Reserve, Ibitipoca Biological Reserve, Augusto Ruschi Biological Reserve, Córrego de Areia Farm and Caratinga Biological Station). Supplementation with 5 individuals per year was the best scenario for the Alto Cariri population and the union of nearby fragments + 5% dispersion rate + supplemented with 1 individuals / year was the best scenario for Santa Maria de Jetibá population. The results allow us to glimpse the best investments of conservation resources and will find great value in further discussions on the handling of the remains in order to protect the species.

Keywords: conservation biology, systematic planning, VORTEX, population viability analysis, PVA; modeling; Atlantic.

1. Introduction

Due to the herculean magnitude of the task to conserve biodiversity, we need to devise ways to strategically plan conservation actions (Margules and Pressey, 2000). The identification of priority sites and species are the first stage, but to reconcile conservation goals to the socio-economic opportunities/limitations represents an important complementary step (Margules and Pressey, 2000; Naidoo et al. 2006; Wilson et al. 2007).

Until recently, most studies aim to measure and map biological diversity, not taking into account socio-economic aspects of the implementation of the actions suggested, creating a gap between theory and practice in Conservation Biology (Naidoo et al. 2006). In Brazil, this scenario becomes quite evident, since less than 20% of the national scientific production had socio-economic and political aspects in the objectives (Grelle et al. 2009). Taking into account that Brazil houses a substantial fraction of global biodiversity (Mittermeier et al. 2005; Drummond, 2008), it is reasonable to consider that there is a great demand for conservation studies incorporating costs into the analyses.

Over 10% of the 658 Brazilian mammal species show some risk of extinction (Paglia et al. 2008; IUCN, 2010). Of these, primates stand out with 26 threatened species, the largest number among mammals (Chiarello et al. 2008). Being one of the most threatened *taxa* (Dobson and Lyles, 1989), many studies focus in primates to investigate processes of extinction and conservation (Dobson and Lyles, 1989; Gibbons and Harcourt, 2009). Of the 98 primate species native to Brazil (Chiarello et al. 2008), the northern muriqui, *Brachyteles hypoxanthus* occupies an unfortunate position among the most threatened primates of the Earth (Strier et al. 2006). Moreover, it is an endemic species of Atlantic Forest, one of the most threatened biomes of the planet (Mittermeier et al. 2005)

In this context, efforts that aim to evaluate how different management options improve the viability of the species, and estimate how much it would cost to implement such plans could reduce the gap between theoretical and practical aspects in Atlantic Forest conservation. The purpose of this study was to use viability models to guide the management decisions for northern muriqui populations and incorporate costs for the management options identified as priority actions for the conservation of the northern muriqui.

2. Material and methods

2.1 *Natural history of Brachyteles hypoxanthus*

The genus *Brachyteles* comprises the largest species of Neotropical primates (Aguirre, 1971). Endemic to the Atlantic Forest, their natural distribution ranges from the coastal forests of the Serra do Mar, southeastern Brazil, to semi-deciduous forests inland (Aguirre, 1971). *Brachyteles hypoxantus* is considered herbivorous (Milton, 1984; Carvalho et al. 2004; Talebi et al. 2005), but also consume seeds, pollen, bamboo and ferns (Strier, 1991a). It has diurnal and arboreal behavior (Dib et al. 1997).

They can form groups of more than 50 individuals (Strier, 1993-1994). Males are philopatric and females disperse to new groups at the beginning of the reproductive life, around of six years old (Strier and Ziegler, 2000). The first offspring are born around nine years of age (Strier and Ziegler, 2000; Strier, 2005). There is only one reproductive event per year where one individual is born. The females show parental care for approximately three years (Strier, 1993/1994, 1996). The maximum reproductive age for northern muriqui is around 35 years old (Rylands et al. 1998). The mean density is 0.0605 (± 0.0899) individuals per hectare (Strier and Fonseca, 1996/1997).

2.2 *PVA*

The analysis was conducted using the software VORTEX version 9.92 (Lacy et al. 2008). VORTEX is one of the modeling tools most used in population viability analysis (Lindenmayer et al. 1995). VORTEX uses Monte Carlo simulations to simulate the effects of deterministic and stochastic forces on population dynamics, reproducing a typical sequence of events in the life cycle of organisms (Lacy 1993, 2000a). Lacy (1993, 2000b) present a detailed description of the software.

2.3 *Simulation and scenarios*

The thirteen extant northern muriqui populations confirmed in nature (see Mendes et al. 2005b) were analyzed. The demographic data used as input to create the viability models were compiled from the available published scientific literature on northern muriqui (Strier, 1991b; Strier, 1993/1994; Strier, 2000; Strier, 2005; Strier et al. 2006; Rylands et al. 1998; Brito and Grelle, 2006; Coutinho, 2007; Brito et al. 2008). To quantify the benefits (*B*) for each management scenarios tested, we use the length of the vectors of each simulated scenarios in the Cartesian plane (where the axes was the probability of extinction - *PE* and heterozygosity - *He*). The equation employed was (Batschelet, 1978):

$$B = \sqrt{PE^2 + He^2}$$

The values of *B* for each scenario was plotted against its costs (see section 2.4), allowing selection of the best scenarios considering the costs. The critical values used for the determination of population viability were *PE* = 0.01 and *He* = 0.9. Populations were modeled for a time frame of 50 generations and were assumed a temporal extension of 10 years for every management programs simulated.

For the long-term viable populations (see Chapter 2), only scenarios simulating harvesting of individuals were employed (1 and 5 females per year) in order to assess the population ability to be source of individuals in population management programs. The inability of Fazenda Esmeralda population to persist in long term made it to be considered as a source of animals too. Regarding the populations with some management demand, based on suggested priority actions for the conservation of *Brachyteles hypoxantyhus* (Mendes et al. 2005b), the following scenarios were created: (1) addition of habitat at 2% per year, (2) addition of habitat at 5% per year, (3) translocation of a female per year, (4) translocation of 5 females per year, (5) supplement 1 female per year from captive populations, (6) supplementation of 5 females per year from captive populations, (7) translocation of 1 female per year + habitat incorporation at 5% per year, (8) translocation of 1 female per year + habitat incorporating at 2% per year; (9) translocation of 5 females per year + habitat incorporation at 2% per year. In Santa Maria de Jetibá, where the population is scattered in 13 subpopulations (Mendes et al. 2005a; Coutinho 2007), the scenarios were: (1) union of closest subpopulations; (2) union of closest subpopulation + 5% dispersion and (3) union of closest subpopulations + 5% dispersion + supplementation of 1 female per year.

Cost estimatives

Estimates of costs for translocation and the maintenance of captive populations were obtained from previous experience of several muriqui management programs (Fabiano Melo, unpublished data; Alcides Pissinatti, unpublished data). Thus, we obtained an estimate of US$ 281065.10 for the translocation of one individual per year; US$

979,289.94 for the translocation of 5 individuals per year; US$ 331,621.00 for supplementation with one individual per year from captive population; US$ 823,005.92 supplementation with five individuals per year from captive population. There is an estimated cost of US$ 1329.47 per hectare recovered in Brazil (Duca et al. 2009). Finally, for combined scenarios (e.g. translocation of 1 female per year + habitat incorporating at 2% per year), the respective costs was combined too.

3. Results

The models that assessed the potential of the five viable populations (Mata Escura Forest Biological Reserve, Rio Doce State Park, Caparaó National Park, Serra do Brigadeiro State Park, Itatiaia National Park) as source of individuals for population management programs indicate that only Caparaó National Park supports a harvest rate of five females annually over ten years, while the others only supported the harvest of one female (table 1). Thus, adding up the three individuals in the unviable Fazenda Esmeralda population plus the individuals harvested from the donor populations, it would be possible to obtain 83 individuals for translocations or reintroductions in a decade.

For populations demanding management interventions, incorporating the costs in analysis allow us to observe that the best management options were not the most costly in most cases (fig. 1, Appendix A). The results for the seven populations in demand for management interventions follow:

Alto Cariri National Park

The best result for the Alto Cariri population was to combine incorporation of habitat at 2% per year and the supplementation of 5 females per year (fig. 1). However, considering the costs involved, female supplementation at 5 individuals per year (through captive populations) decrease the benefits in 0.34%, but costs decrease 84.22%. Supplementation through translocation of 5 females (from donor populations) produces identical benefits, but costs are 15.96% higher than supplementation through captive populations.

RPPN Mata do Sossego

The translocation of 1 female per year and habitat incorporation at 5% per year was the scenario with the best results (fig. 1). However, just incorporating habitat saves 65.18% of management costs with only 0.61% of decrease in benefits, indicating that such rate of habitat management yields the best resources investments to the conservation of this population.

Fazenda Córrego de Areia

The translocation of one female per year and habitat incorporation at 5% per year yielded the best results for this population (fig. 1). However, when we consider the costs, the habitat management alone reduces costs by 68.18% in exchange of a 0.05% decrease in benefits.

Ibitipoca State Park

Habitat incorporation at 5% per year was the best scenario for the management of Ibitipoca population (fig. 1). The incorporation of less habitat (2% per year) generates a reduction of 65.18% in costs in exchange of 2.03% of benefits.

Augusto Ruschi Biological Reserve

The combined scenario of supplementation of 1 female per year and the incorporation of habitat at 5% per year generated the best results (fig. 1). Limiting actions to habitat management reduces 1.37% of the benefits and 8.62% of the costs.

Caratinga Biological Station

The incorporation of habitat at 5% per year generated the best benefit for the management of Caratinga population (fig. 1). The scenario for the translocation of 1 female per year + 5% of habitat per year was the only one that yielded similar benefits, but it is 26.04 % more expensive.

Santa Maria de Jetibá forest fragments

For the northern muriqui population in Santa Maria de Jetibá, only the scenario for the connection of close populations associated with a 5% increase in the dispersal rate between subpopulations and the supplementation of 1 female per year generated benefits (fig. 1). These outputs indicate that the effort to conserve this population should involve the complex task of combining different action fronts.

Table 1: Outputs from simulations in VORTEX to assess potential source populations of northern muriqui individuals for management programs. In the baseline scenario the population was simulated with no one management intervention.

Population	**Baseline**		**Harvest 1 female/year/10 year**		**Harvest 5 female/year/10 year**	
	PE	*He*	*PE*	*He*	*PE*	*He*
Mata Escura Biological Reserve	0.0	0.9426	0.0	0.9350	***0.95***	***0.6934***
Rio Doce State Park	0.0	0.9355	0.0	0.9269	***0.95***	***0.7235***
Caparaó National Park	0.0	0.9564	0.0	0.9553	0.00	0.949
Serra do Brigadeiro National Park	0.0	0.9205	0.0	0.9203	***0.95***	***0.7235***
Itatiaia National Park	0.0	0.9484	0.0	0.9412	***0.94***	***0.7127***
Fazenda Esmeralda	***1.0***	***0.0000***	***1.0***	***0.0000***	***1.00***	***0.0000***

*Values in italics indicate non-viable scenarios (i.e. *PE* > 0.05 and/or *He* < 0.90; details in Material and Methods)

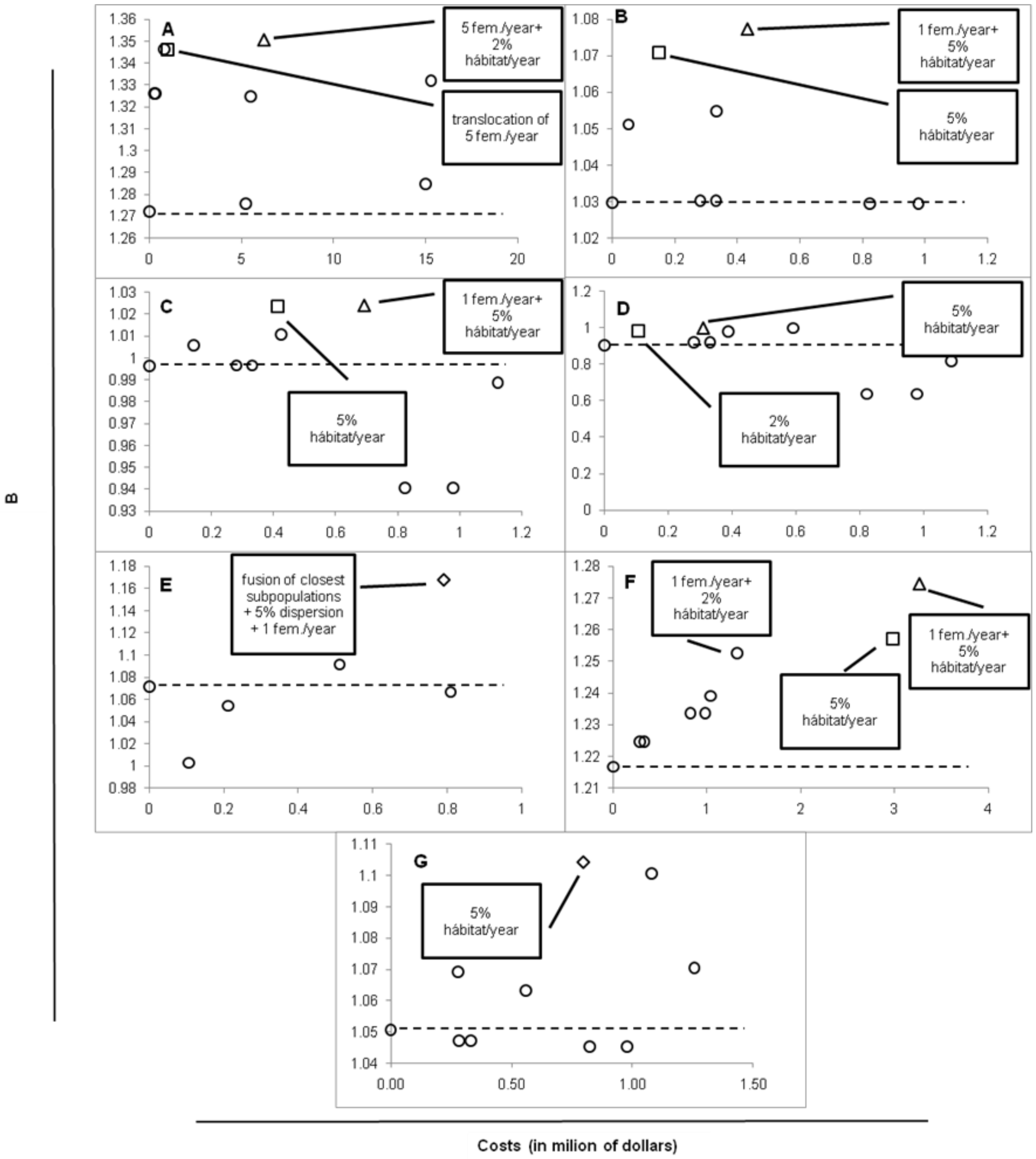

Figure 1: Results for the costs vs. benefits analysis (detailed results in Appendix A). In the *Y* axes are the benefits (*B*, see section 2.3 in Materials and methods), and in the *X* axes are the costs of every management scenario simulated. The dotted line indicates the baseline scenario (i.e. simulation without management intervention and, therefore, without costs). A = Alto Cariri National Park; B = Mata do Sossego Private Natural Heritage Reserve; C = Fazenda Córrego de Areia; D = Ibitipoca State Park; E = Santa Maria de Jetibá; F = Augusto Ruschi Biological Reserve; G = Caratinga Biological Station. Δ = best case scenario ignoring the costs; □ = best case scenario considering the costs; ◊ = best case scenario, with or without costs.

4. Discussion

Of the three management actions employed in the construction of scenarios (supplementation through translocation, supplementation through captive populations and incorporation of habitat), habitat management was able to yield the best combinations between costs and benefits , corroborating Mendes et al. (2005b), who considered the management of the habitat among the most important strategy for northern muriqui conservation.

Regarding genetic diversity, habitat management yielded a greater benefit in smaller areas (e.g. Mata do Sossego Private Natural Heritage Reserve), whereas the supplementation of individuals yielded the greatest benefits in populations located in larger areas (e.g. Alto Cariri National Park). Additional studies could contribute to a better exploitation of this apparent pattern and may yield important insights about the existence of thresholds for the efficiency of different management options as a function of intervention area extension. For example, Duca et al. (2009), studying the cost-effectiveness for the management of white-banded tanager, *Neothraupis fasciata* (Aves: Thraupidae) in protected areas of the Brazilian Cerrado, found that the size of the area is an important factor in the efficiency of management options analyzed (fire management and nest protection), where the management of fire is the better option in smaller areas (i.e. <1000 ha) and both options exhibit the same efficiency in large areas (i.e. > 4000 ha).

The incorporation of costs has shown that often the distance between an efficient and an inefficient conservation plan is narrow (Polasky et al. 2001; Polasky et al. 2005). As a example, in the Willamette River Basin (Oregon, United States), Polasky et al. (2005) found that giving up 25% of the goals aimed to the conservation of this landscape causes just a 7% decrease in local economy profits. However, if the option is not to give up of such 25%, the local economy return can be reduced in something around 70%. In this sense, the results presented here helped confirm that for six of the seven populations in demand for management interventions, the best scenario not always is the most expensive. As a result, not considering costs in ranking management options could lead to unnecessary expenses of meager conservation funds. For example, for the northern muriqui population at Alto Cariri National Park, if costs area ignored, a manager could (naively) choose the third-best scenario and spend 41% more resources and have less results regarding population recovery than he would achieve if he chose the best scenario.

There are projects for northern muriqui conservation that have been implemented and allow us to glimpse the feasibility of management options simulated in this study. Pontual and Boubli (2005) reported that in Caratinga, 30 hectares of forests were planted in six months' work, with the engagement of local landowners. With respect to the translocation of individuals, the first experiences were successful (Fernando Melo, unpublished data), contributing to the refinement of this management option. Pissinati (2005) reports the success of pioneering initiatives to reproduce the northern muriqui in captivity, through the commitment of the institutions involved. Thus, this study contributes by providing theoretical basis for optimizing decision making and will be important for further discussions regarding management to northern muriqui conservation.

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7. Appendix A

Complete results of the VORTEX simulations for the seven northern muriqui populations that exhibits demands for management.

Population	**Scenarios**	**US$**	***PP***	***He***	***B***	**Best scenarios**
Alto Cariri National Park	Base-line	0.00	0.99	0.80	1.272	8º
	Translocation of 1 female/year	281065.10	1.00	0.87	1.326	4º
	Translocation of 5 females/year	979289.94	1.00	0.90	1.346	2º
		331621.00	1.00	0.87	1.326	4º

Alto Cariri National Park	Suplementation of 1 captive female/year					
	Suplementation of 5 captive females/year	823005.92	1.00	0.90	1.346	2º *
	Incorporation of habitat at 2%/year	5228417.90	0.99	0.81	1.276	7º
	Incorporation of habitat at 5%/year	15014641.04	0.99	0.81	1.285	6º
	Translocation of 1 female/year + habitat incorporation at 5%/year	15295706.14	1.00	0.89	1.332	3º
	Translocation of 1 female/year + habitat incorporation at 2%/year	5509483.00	0.99	0.88	1.325	5º
	Translocation of 5 females/year + habitat incorporation at 2%/year	6207707.84	1.00	0.91	1.351	1º
	Base-line	0.00	1.00	0.25	1.03	7º
	Translocation of 1 fema-	281065.10	1.00	0.25	1.03	6º

Mata do Sossego Particular Reserve of Natural Heritage	le/year					
	Translocation of 5 females/year	979289.94	1.00	0.25	1.03	8º
	Suplementation of 1 captive female/year	331621.00	1.00	0.25	1.03	6º
	Suplementation of 5 captive females/year	823005.92	1.00	0.25	1.03	8º
	Incorporation of habitat at 2%/year	52285.51	1.00	0.32	1.051	4º
	Incorporation of habitat at 5%/year	150145.08	1.00	0.38	1.071	2º *
	Translocation of 1 female/year + habitat incorporation at 5%/year	431210.18	1.00	0.40	1.077	1º
	Translocation of 1 female/year + habitat incorporation at 2%/year	333350.61	1.00	0.34	1.055	3º

Mata do Sossego Particular Reserve of Natural Heritage	Translocation of 5 females/year + habitat incorporation at 2%/year	1031575.45	1.00	0.30	1.045	5º
Córrego de Areia Farm	Base-line	0.00	0.99	0.09	0.996	6º
	Translocation of 1 female/year	281065.10	0.99	0.10	0.997	5º
	Translocation of 5 females/year	979289.94	0.94	0.09	0.941	8º
	Suplementation of 1 captive female/year	331621.00	0.99	0.10	0.997	5º
	Suplementation of 5 captive females/year	823005.92	0.94	0.09	0.941	8º
	Incorporation of habitat at 2%/year	143486.71	1.00	0.14	1.006	4º
	Incorporation of habitat at 5%/year	412063.37	1.00	0.22	1.024	2º *
	Translocation of 1 female/year + habitat incorpo-	693128.47	1.00	0.22	1.024	1º

	ration at 5%/year					
Córrego de Areia Farm	Translocation of 1 female/year + habitat incorporation at 2%/year	424551.81	1.00	0.15	1.011	3º
	Translocation of 5 females/year + habitat incorporation at 2%/year	1122776.65	0.98	0.15	0.989	7º
Ibitipoca State Park	Base-line	0.00	0.90	0.05	0.905	6º
	Translocation of 1 female/year	281065.10	0.92	0.04	0.921	5º
	Translocation of 5 females/year	979289.94	0.63	0.06	0.637	8º
	Suplementation of 1 captive female/year	331621.00	0.92	0.04	0.921	5º
	Suplementation of 5 captive females/year	823005.92	0.63	0.06	0.637	8º
	Incorporation of habitat at 2%/year	108059.36	0.98	0.07	0.981	4º *

Ibitipoca State Park	Incorporation of habitat at 5%/year	310304.26	0.99	0.13	1.001	1º
	Translocation of 1 female/year + habitat incorporation at 5%/year	591369.36	0.99	0.13	0.999	2º
	Translocation of 1 female/year + habitat incorporation at 2%/year	389124.46	0.98	0.09	0.982	3º
	Translocation of 5 females/year + habitat incorporation at 2%/year	1087349.30	0.82	0.07	0.819	7º
Santa Maria de Jetibá	Base-line	0.00	1.00	0.40	1.072	3º
	Merging the closest	212219.20	1.00	0.33	1.054	5º
	Union of all	809085.70	1.00	0.37	1.067	4º
	dispertion at 5%	106109.60	0.96	0.28	1.003	6º
	Merging the closest + dispertion at 5%	510652.45	1.00	0.44	1.092	2º
	Merging the closest + dispertion at 5% + 1 female/year	791717.55	1.00	0.60	1.168	1º *
	Base-line	0.00	1.00	0.69	1.217	8º
	Translocation of 1 female/year	281065.10	1.00	0.71	1.225	7º

Augusto Ruschi Biological Reserve	Translocation of 5 females/year	979289.94	1.00	0.72	1.234	5º
	Suplementation of 1 captive female/year	331621.00	1.00	0.71	1.225	7º
	Suplementation of 5 captive females/year	823005.92	1.00	0.72	1.234	5º
	Incorporation of habitat at 2%/year	1037844.73	1.00	0.73	1.239	4º
	Incorporation of habitat at 5%/year	2980406.45	1.00	0.76	1.257	2º *
	Translocation of 1 female/year + habitat incorporation at 5%/year	3261471.55	1.00	0.79	1.275	1º
	Translocation of 1 female/year + habitat incorporation at 2%/year	1318909.83	1.00	0.75	1.253	3º
	Translocation of 5 females/year + habitat incorporation at 2%/year	2017134.67	1.00	0.72	1.233	6º

Caratinga Biological Station	Base-line	0.00	1.00	0.32	1.051	6º
	Translocation of 1 female/year	281065.10	1.00	0.31	1.047	7º
	Translocation of 5 females/year	979289.94	1.00	0.30	1.045	8º
	Suplementation of 1 captive female/year	331621.00	1.00	0.31	1.047	7º
	Suplementation of 5 captive females/year	823005.92	1.00	0.30	1.045	8º
	Incorporation of habitat at 2%/year	277980.62	1.00	0.38	1.069	3º
	Incorporation of habitat at 5%/year	798275.78	1.00	0.47	1.104	1º *
	Translocation of 1 female/year + habitat incorporation at 5%/year	1079340.88	1.00	0.46	1.101	2º

Caratinga Biological Station	Translocation of 1 female/year + habitat incorporation at 2%/year	559045.72	1.00	0.36	1.063	5º
	Translocation of 5 females/year + habitat incorporation at 2%/year	1257270.56	1.00	0.38	1.07	4º

*The best scenarios considering the costs.

CONCLUSÃO GERAL

Estimativas recentes relatam que os remanescentes naturais da Mata Atlântica somam algo entre 11% e 16% de sua cobertura vegetal original (Ribeiro, J. P. Metzger, Alexandre Camargo Martensen, Ponzoni, & Hirota, 2009). Tão grave quanto à constatação de que a maior parte deste bioma foi devastada é a situação em que se encontra a fração remanescente: mais de 80% trata-se de fragmentos menores que 50 ha, separados a uma distância média de quase 1,5 km (Ribeiro, J. P. Metzger, Alexandre Camargo Martensen, Ponzoni, & Hirota, 2009). Mas a pressão antrópica da perda de hábitat não é uma característica exclusiva da Mata Atlântica, ocorrendo também em vários outros biomas ao redor do planeta (Myers 1988; Mittermeier et al. 1998; Myers et al. 2000). Como consequência direta, uma das maiores ameaças às espécies nas “listas vermelhas” são a fragmentação e a perda de hábitat (IUCN 2010). Este cenário permite vislumbrar a relevância e o potencial prático do arcabouço teórico da biologia de populações, fornecendo a base teórica para a elaboração de soluções eficientes na conservação da biodiversidade num mundo onde os remanescentes naturais das espécies tratam-se de populações esparsas numa paisagem fragmentada.

Um dos conceitos teóricos essenciais para quaisquer programas de conservação é o tamanho mínimo para a viabilidade de uma população (Reed et al. 2003; Traill et al. 2007). De acordo com a teoria, em paisagens altamente fragmentadas (como na Mata Atlântica), a progressiva transformação da matriz florestal nativa em sistemas criados pelo homem colapsa a área de hábitat mínima para a manutenção de uma população viável (Metzger & Dècamps 1997; Uezu et al. 2005; Martensen et al. 2008). Nessas situações, a intuição nos faz conjecturar em desfavor das espécies de grande porte (e que demandam extensas áreas de hábitat), como o muriqui-do-norte na Mata Atlântica. Os resultados dos 223 cenários simulados no Capítulo 2 indicam que apenas quatro das treze populações do muriqui-do-norte são viáveis em longo prazo, corroborando a ideia de que a fragmentação/perda de hábitat, na intensidade em que ocorreu na Mata Atlântica, imprime um forte efeito negativo sobre as espécies.

Nas situações em que as áreas de hábitat disponíveis para uma espécie se encontram dispersas na matriz da paisagem, a conectividade entre tais porções remanescentes de habitat torna-se um fator crítico (Martensen et al. 2008; Metzger & Dècamps 1997;

Uezu et al. 2005). Mas uma população mínima viável comporta-se da mesma forma quando numa configuração meta-populacional? Esta foi a questão central do Capítulo 1 e as projeções dos 13 cenários simulados indicam que a resposta é *não*. Achados similares foram publicados para outras espécies (Lindenmayer & Lacy 1995a; Lacy & Lindenmayer 1995; Lindenmmayer & Lacy 1995b; Brito & Fonseca 2007) e, em seu conjunto, pavimentam o caminho para que esforços empíricos sejam destinados a testar mais detalhadamente esta negativa.

Se a deterioração biológica da Mata Atlântica é, por si só, um problema de complexa solução, a incorporação da dimensão socioeconômica aos esforços de conservação agrava ainda mais a situação. Conforme exposto por Margules & Pressey (2000), desconsiderar a dinâmica das sociedades nas quais as ações de conservação serão implementadas é um forte convite ao fracasso. O terceiro capítulo desta dissertação buscou contribuir neste sentido, através da incorporação de uma dimensão econômica aos esforços de conservação do muriqui-do-norte. As diferentes combinações entre manejo de hábitat, suplementação de indivíduos e suas respectivas estimativas de custos renderam a simulação de *68* cenários de manejo para os treze remanescentes populacionais confirmados na natureza. Ao logo dos resultados apresentados no referido capítulo, observou-se que o manejo de hábitat foi a intervenção que mais frequentemente apareceu como o melhor investimento para a conservação das populações. Em paisagens como as da Mata Atlântica, esta é uma tarefa que inevitavelmente demanda por um forte engajamento de proprietários rurais para o estabelecimento de um complexo de corredores florestais. Assim, se os resultados obtidos neste capítulo encontrarem valor junto a gestores e tomadores de decisão, espera-se que o manejo para a conservação do muriqui-do-norte represente também uma contribuição inerente para o manejo e conservação da própria fitofisionomia da Mata Atlântica.

O panorama geral das informações produzidas ao longo dos capítulos permitiu realçar os principais pontos para o embasamento teórico para o manejo populacional do muriqui-do-norte:

- configurações metapopulacionais prejudicam a dinâmica da população mínima viável para a espécie;

- a conectividade metapopulacional mitiga apenas parcialmente os efeitos negativos da subdivisão populacional, havendo um limiar a partir do qual eventos de migração exercem um forte efeito negativo;

- estratégias de incorporação e manejo de hábitat devem ser priorizadas em investimentos de pesquisa e de recursos financeiros, tendo em vista permitir que a maior parte das populações que demandam por intervenção sejam efetivamente manejadas;

- as cinco populações estimadas como viáveis em longo prazo (Reserva Biológica Mata Escura, Parque Estadual do Rio Doce, Parque Nacional do Caparaó, Parque Estadual Serra do Brigadeiro e Parque Nacional de Itatiaia) são atualmente os principais remanescentes para a conservação da espécie;

- esforços de pesquisa sobre translocação e reintrodução da espécie focal poderiam contribuir para que novas populações sejam formadas em áreas que poderiam suportar populações viáveis.

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