Range shifts under climate change and the role of protected areas for armadillos and anteaters

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ABSTRACT

The planning of protected areas takes into consideration the current distributions of target species, disregarding possible future range shifts under climate change. This study tested potential shifts in the distribution of three xenarthran species (the giant anteater, Myrmecophaga tridactyla; the giant armadillo, Priodontes maximus; and the three-banded armadillo, Tolypeutes tricinctus) due to the effects of climate change under different dispersal scenarios. In addition, we tested the adequacy of the current Brazilian protected areas in preserving these species under future climate changes (future distribution for the year 2050), under full and null dispersal scenarios. The final occurrence maps were then confronted with two Brazilian reserve systems for the year 2010: one consisting of reserves for integral protection only (IPCU), and one that included also reserves for sustainable use and indigenous reserves (SUCU). In the IPCU system, none of the species have or will have > 10% of their distributions protected, under neither dispersal scenario. Including SUCU, however, M. tridactyla and P. maximus already are, and will still be, adequately protected in the future. T. tricinctus, which is endemic to the Brazilian semi-arid, will not reach the 10% goal under neither dispersal scenario, even under the SUCU system. We stress the need for management actions in the northeastern and central parts of Brazil. Opportunities for the creation of more reserves are highlighted, considering the potential range shifts and the natural habitats still present in these regions.

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1. Introduction

Protected areas for nature conservation are commonly planned according to political and economic criteria, resulting in a reserve system that often does not adequately represent the most important targets of biodiversity (Pressey, 1994; Rodrigues et al., 2004a). The consequence is the lack of an integrated system of protected areas, planned for filling gaps within the current conservation efforts, and considering the future survival of the species to be protected (Machado et al., 2004; Pressey et al., 2007; Gibson et al., 2010; Cianfrani et al., 2011). Nowadays, the easy access to Geographic Information Systems allows the broad use of tools and methods to assess the status of the currently protected areas, thus helping in the planning of new reserves (Peterson et al., 2002a; Scott et al., 1987).

Gap assessments are one of these methods, consisting on identifying components of biodiversity that are unrepresented in the existing reserve system (Scott et al., 1993). Current or future distribution maps of the target species are confronted with a map of protected areas, enabling the quantification of the current protection for those species. When the current reserves represent a species’ range poorly (i.e. the protected area is smaller than that established as a protection threshold), there is a gap in the conservation of a species. Once the gaps are identified, priority areas for conservation and management are appointed, by selecting areas that optimally represent those species with gaps in their protection, and that complement the existing reserve system (Scott et al., 1987, 1993). The first step in this assessment is to determine reliably the geographic ranges of the target species. This information, however, is lacking for many species, including groups of conservation concern. Maps that represent the ranges of the target species thus have to be produced using modeling procedures currently widespread (Peterson et al., 2000; Allen et al., 2001; Anderson et al., 2003; Rushton et al., 2004; Vital, 2005; Catullo et al., 2008).

The rapid rate of global climatic change, however, may render the efforts of filling current gaps ineffective (Araújo et al., 2004; Hannah et al., 2007; Pressey et al., 2007; Wiens et al., 2011). Modeling tools are thus also useful in helping to predict and prepare for future events, increasing the cost-effectiveness of management actions. In the face of climate change, species are expected to
behave in idiosyncratic ways, preferably undergoing range shifts (Hannah et al., 2008; Schweiger et al., 2008; Virkala et al., 2008). Range shifts, however, will not always be possible, and accounting for the species’ different dispersal abilities, for the effects of land use change, and for the synergy among other anthropogenic factors is a necessary step for the assessment of how the species will succeed in adapting to a changing climate (Sala et al., 2000; Laurance and Williamson, 2001; Jetz et al., 2007; Lindenmayer et al., 2010).

Armadillos and anteaters (Mammalia: Xenarthra) form a threatened and still poorly known group of mammals. They play an important evolutionary and ecological role in the Neotropical region, as they are key species in the cycling of nutrients in the ecosystems in which they occur (Eisenberg and Thorington, 1973; Redford, 1985), and important prey species for the top predators in the food-chain (Marinho-Filho et al., 2002). According to the Brazilian list of endangered mammals (Chiarello et al., 2008), the systems in which they occur (Eisenberg and Thorington, 1973; Redford, 1985), and important prey species for the top predators in the food-chain (Marinho-Filho et al., 2002). According to the Brazilian list of endangered mammals (Chiarello et al., 2008), the giant armadillo (Priodontes maximus Kerr, 1792), the three-banded armadillo (Tolypeutes tricinctus Linnaeus, 1758) and the giant anteater (Myrmecophaga tridactyla Linnaeus, 1758) are classified as vulnerable. However, up to the present study there has been little information about the precise geographic distribution of these species. Only one study has modeled the distribution for the Dasypodidae family (armadillos; Anacleto et al., 2006).

The objective of this study was to model the distribution of the endangered xenarthrans (M. tridactyla, P. maximus and T. tricinctus) in the future, and to identify gaps in the protection of these species in the current Brazilian reserve system, assessing potential future gaps in the face of climate change. We examined two range shifts scenarios: in one, the species are considered able to migrate freely (full dispersal scenario), whereas in the other, they are constrained in their ability to shift their distributions (null dispersal scenario) (Marini et al., 2009a; Pearson, 2006; Peterson et al., 2002b; Gibson et al., 2010).

2. Materials and methods

2.1. Specimen and environmental data

The specimen data were composed of less precise data compiled from the literature, and more precise data obtained from research projects (Fig. 1). The data from the literature were gathered based on the correlation by Anacleto et al. (2006) as a starting point. Data from zoological collections and occurrence lists from the literature were included (see Appendix A). Localities outside Brazil were found by consulting the Global Gazettee v. 2.2 (http://www.fallingrain.com/world), and data from municipalities and localities that lacked the exact geodesic coordinates of occurrence were referenced with the aid of Google Earth 5.1.2.1588. These data are part of the least precise database. All of them had their positions corrected on Google Earth so as to fall on native vegetation-covered areas on the proximity of the original locality coordinates, which tend to fall on the municipalities’ central towns. Data that derived from projects conducted by the Universidade de Brasília, by the Jaguar Conservation Fund (JCF), and local researchers’ personal observations were also compiled. This is the most precise part of the database, including GPS recorded point localities, and presenting an average accuracy of 10 m. Most GPS occurrence points for M. tridactyla and P. maximus, resulting from intensive field sampling, were highly concentrated in the sampled areas. To deal with this problem, a grid was created (10 km resolution), and only one occurrence point among those that fell within the same grid cell was maintained. This way, the high spatial autocorrelation presented by those clustered points was minimized (Bini et al., 2006; Guisan and Thuiller, 2000).

The climatic variables include precipitation and temperature data for both the present day and for the climate change scenario (described below). All environmental data were in 5’ resolution (cell side of 10 km). In order to exclude variables that were too correlated with each other, a principal component analysis was performed for the present day data in the program R 2.12.0 (R Development Core Team, 2010). The same selected variables were also used in the analysis of the future scenario, and they were: altitude, annual mean temperature, mean diurnal range, maximum temperature of warmest month, minimum temperature of coldest month, temperature annual range, mean temperature of warmest quarter, mean temperature of coldest quarter, annual precipitation, precipitation seasonality (coefficient of variation), precipitation of driest quarter, precipitation of warmest quarter, and precipitation of coldest quarter. The climatic parameters were obtained from the WorldClim bioclimatic database (Hijmans et al., 2005; http://www.worldclim.org), and the altitude variable, from the SRTM spatial mission project http://www2.jpl.nasa.gov/srtm/data.html.

The global climatic models used to produce the future climatic variables, came from three sources (CCcm – Canadian Center for climate modeling and analysis; Hadcm – Hadley Center Coupled Model – United Kingdom; Csiro – Commonwealth Scientific and Industrial Research Organization – Australia). In this study, we took the average of the mean, minimum and maximum temperatures, and the precipitation from all three models under one emission scenario (scenario A2a, that describes a highly heterogeneous future world characterized by a high rate of population growth, increased energy use, land-use changes and slow technological change; IPCC, 2001), and generated the other bioclimatic variables, by using a script provided in the WorldClim website.

A 10 km resolution was considered an adequate measure for several reasons. Results of spatial models like the one here produced are highly scale dependent (Guisan and Thuiller, 2005), and since a great portion of our database was composed of relatively imprecise data, the choice for the adequate resolution had to be guided by this limitation. Also, mammals are species that present comparatively large home range areas, and it is argued that if a species occurs in a 10 km resolution cell, probably all the habitat patches present in that cell will be inhabited (Scott et al., 1987). Moreover, according to Allen et al. (2001), the resolution must either match the size of a home range area of an individual, or be larger. In a very fine resolution, there is too much source of variation for species that perceive the landscape in a coarser manner. Suitable cells that are too small can be classified as absence for a certain species due to effects of intra-specific interactions, or the lack of saturation of the population. In the latter case, not all suitable patches are inhabited due to a low population density in that area. Finally, it is generally considered that geographic ranges at a coarse scale are dominantly defined by bioclimatic variables, which are the factors we considered in this study (Pearson and Dawson, 2003).

2.2. Species-distribution modeling

We used the maximum entropy algorithm (Maxent) for the distribution modeling, which is based on the principle that the species’ distribution must be the most uniform (random) possible, subject only to certain restrictions that represent the relationship between the biological data and the environmental variables (Phillips et al., 2004, 2006). This procedure requires the providing of presence data only. Absence data are generated by randomly selecting “pseudo-absence” points or the background of the area (Anderson et al., 2002, 2003). The resulting model is a map of suitability, which represents the potential distribution of a species (Peterson and Vieglais, 2001). We validated our resulting models
by randomly splitting the data into training data (70% of the data) and test data (30% of the data), by the cross-validation method. We then classified the output map from Maxent by the 10% fixed cumulative value threshold. This is a relatively restrictive threshold, which minimizes the commission error, because in gap assessments and in the definition of priority areas for conservation, the cost of false-positive errors is great (Liu et al., 2005; Loiselle et al., 2003).

For the creation of a final presence/absence map, the average of the 10 replicates generated by the model was then classified, based on the threshold value selected above. As a final validation for the models, we performed a one-tailed binomial test to assess whether the number of points correctly classified by the model was significantly greater than random (Anderson et al., 2002). The map calculations and classifications were performed for the models for all three species in the present and in the future scenarios, using the program ArcGIS v. 9.3.1 (Environmental Systems Research Institute, 2009). The binomial test was performed in the program R 2.12.0 (R Development Core Team, 2010).

2.3. Gap analysis

For the gap analysis for the three xenarthran species, we used a map of the Brazilian reserve system (IBAMA, 2010; Fig. 2). Because we would like to remain conservative in the identification of suitable areas for the conservation and maintenance of mammal populations, and due to the empirical fact that only large areas are able to maintain viable populations of most mammal species, as must be the case for xenarthrans, which have great home range area requirements (300–900 ha; Medri and Mourão, 2005), we selected only reserves over 30,000 ha (Fig. 2).

All gap assessments were made including two groups of protected areas: Integral Protection Conservation Units (IPCU) only, and all reserves, including also Sustainable Use Conservation Units (SUCU; Fig. 2). IPCU are very restrictive categories defined by the Brazilian legislation, aimed at biological protection, research activities, and controlled visitation. SUCU, on the other hand, allow human occupation, such as indigenous and local communities, committed to the sustainable use of their lands. We included indigenous reserves in the analyses for the SUCU system, since they may also contribute with the maintenance of biodiversity. Surveillance efforts of the SUCU, however, are not entirely reliable in the country, and the adequacy of these reserves for biological conservation is an important issue, since many present high rates of deforestation (Peres and Zimmerman, 2001; Zimmerman et al., 2001). Therefore, we selected only those reserves that still presented at least 90% of their native vegetation up to the year 2008, based on the maps of remaining vegetation from the PRODES/INPE (http://www.obt.inpe.br/prodes/index.html) and the CSR/IBAMA (http://siscom.ibama.gov.br/monitorabiomas/) biome monitoring projects.

We thus confronted the maps of potential distribution modeled for the present and for the future with the map of these two Brazilian reserve systems. The procedure was the one chosen by Catullo et al. (2008), and Marini et al. (2009b), in which the reserves that touch a cell classified as “presence” for a given species are selected. The sum of the area of these reserves is therefore the proportion of the total area of distribution of a species that is considered protected. A threshold of protection of 10% was proposed by Rodrigues et al. (2004b) for species with large geographic ranges (greater than 250,000 km²), as is the case here. Therefore, if a species is not represented by any protected area, it presents a full gap. If its representation is lower than the protection threshold, it presents a partial gap. But if its representation is equal to or higher than that indicated by the chosen threshold, the species is considered adequately protected (Catullo et al., 2008).

We performed the gap analysis for the future taking into account two dispersal scenarios. In the full dispersal scenario, the species were able to fully disperse, and change their geographic ranges according to the suitability maps for the year 2050. In the null dispersal scenario, the species were considered to be unable to disperse in the landscape, and their distributions would be limited to the area of overlap between their present and future distributions (Marini et al., 2009a; Peterson et al., 2002b). The spatial analyses were performed using the program ArcGIS v. 9.3.1 (Environmental Systems Research Institute, 2009).
We generated a map of priority areas for the conservation of the species, by extracting, from their modeled ranges under the less optimistic null dispersal scenario, the regions that still present remaining native vegetation, and are not currently protected by any kind of protection area. The map of remaining native vegetation of Brazil was the same used to select the SUCU reserves, as described above. Since *M. tridactyla* and *P. maximus* occur mainly in the same regions, we performed this analysis considering their range overlap. For *T. tricinctus*, the priority areas were indicated considering its distribution alone, since the overlap with the two other species’ ranges is minimal.

3. Results

3.1. Niche modeling

The amount of occurrence points for each species in each Brazilian biome is relevant for the quality of the final models, and is presented in Table 1. As the two types of data available (GPS points and locality points) represent the more precise and less precise data, respectively, this information for the total number of points is also informed.

The final continuous maps for each species’ present and future distributions (ESRI grid format), multiplied by the presence/absence maps created based on the classification threshold, allowed the visualization of the suitable areas in the regions classified as “presence” for each species in the present and in the future (Fig. 3). The one-tailed binomial test showed a highly significant difference, for all three species, between the number of points correctly classified by the model and by random for the present distribution (*M. tridactyla*: \(Z = -31.8525; p < 0.0001; n_{\text{correct}} = 199\)/
*P. maximus*: \(Z = -23.87058; p < 0.0001; n_{\text{correct}} = 113\)/
*T. tricinctus*: \(Z = -49.39399; p < 0.0001; n_{\text{correct}} = 81\)) and for the potential future distribution (*M. tridactyla*: \(Z = -19.4693; p < 0.0001; n_{\text{correct}} = 200\)/
*P. maximus*: \(Z = -14.5904; p < 0.0001; n_{\text{correct}} = 108\)/
*T. tricinctus*: \(Z = -15.7799; p < 0.0001; n_{\text{correct}} = 92\)).

3.2. Gap analysis

The gap analysis indicated that in the IPCU system, all three species currently have partial gaps in their protection, based on a 10% protection threshold. Considering also the SUCU system, however, *M. tridactyla* and *P. maximus* are already adequately protected (Table 2).

In 2050, under the full dispersal scenario, *T. tricinctus* will suffer a slight range expansion, while *M. tridactyla* and *P. maximus* will suffer range contractions (Fig. 3). The range expansion for *T. tricinctus* will cause a decrease in its protection. In the IPCU system, none of the species will reach the 10% threshold for adequate protection. Considering all reserves, on the other hand, *M. tridactyla* and *P. maximus* will still be adequately protected, while *T. tricinctus* will still not reach the threshold (Table 3).

Under the null dispersal scenario all species were expected to suffer range contraction (Araújo et al., 2004; Peterson et al., 2002b). The protection of *T. tricinctus* will be enhanced in comparison with the full dispersal scenario, but will still not reach the 10% protection goal, in both reserve systems. The range contraction for *M. tridactyla* and *P. maximus* will also cause these species’ protec-

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Table 1
Occurrence points for the three analyzed species, in groups of more precise GPS points and less precise locality points. Also, total number of points for each Brazilian biome.

<table>
<thead>
<tr>
<th>Species</th>
<th>Amazon</th>
<th>Cerrado</th>
<th>Caatinga</th>
<th>Pantanal</th>
<th>Atlantic Forest</th>
<th>Pampa</th>
<th>GPS points</th>
<th>Localities</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Myrmecophaga tridactyla</em></td>
<td>94</td>
<td>95</td>
<td>6</td>
<td>16</td>
<td>39</td>
<td>2</td>
<td>84</td>
<td>187</td>
<td>271</td>
</tr>
<tr>
<td><em>Priodontes maximus</em></td>
<td>90</td>
<td>54</td>
<td>2</td>
<td>6</td>
<td>39</td>
<td>2</td>
<td>39</td>
<td>103</td>
<td>142</td>
</tr>
<tr>
<td><em>Tolypeutes tricinctus</em></td>
<td>0</td>
<td>21</td>
<td>75</td>
<td>0</td>
<td>4</td>
<td>0</td>
<td>14</td>
<td>86</td>
<td>100</td>
</tr>
</tbody>
</table>

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Fig. 2. Reserves larger than 30,000 ha divided in Integral Protection Conservation Units (IPCU) and Sustainable Use Conservation Units (SUCU) selected for the gap analyses, and the six Brazilian biomes (IBAMA, 2010).
Table 2

<table>
<thead>
<tr>
<th>Species</th>
<th>Total area</th>
<th>IPCU</th>
<th>IPCU + SUCU</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Protected area</td>
<td>%</td>
</tr>
<tr>
<td>Myrmecophaga tridactyla</td>
<td>5209,473</td>
<td>474,928</td>
<td>8.96</td>
</tr>
<tr>
<td>Priodontes maximus</td>
<td>5219,811</td>
<td>440,602</td>
<td>8.44</td>
</tr>
<tr>
<td>Tolypeutes tricinctus</td>
<td>895,035</td>
<td>26,414</td>
<td>2.95</td>
</tr>
</tbody>
</table>

Table 3

<table>
<thead>
<tr>
<th>Species</th>
<th>2050 full dispersal</th>
<th>2050 null dispersal</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total area</td>
<td>IPCU</td>
</tr>
<tr>
<td>Myrmecophaga tridactyla</td>
<td>4900,990</td>
<td>467,217</td>
</tr>
<tr>
<td>Priodontes maximus</td>
<td>4705,982</td>
<td>463,115</td>
</tr>
<tr>
<td>Tolypeutes tricinctus</td>
<td>924,885</td>
<td>26,413</td>
</tr>
</tbody>
</table>

Fig. 3. Suitability maps in Brazil generated for Myrmecophaga tridactyla (A), Priodontes maximus (B), and Tolypeutes tricinctus (C) in the year 2000 (1) and 2050 (2).

4. Discussion

M. tridactyla and P. maximus occur mainly in the cerrado biome, in central Brazil (Eisenberg and Thorington, 1973; Eisenberg and Redford, 1999; Wetzel, 1985). This is a highly biodiverse savanna ecosystem, and is considered a biodiversity hotspot (Myers et al., 2000). However, M. tridactyla, in comparison with P. maximus, presents a range that extends southwards, covering a large part of Southeast Brazil. P. maximus has a more northern range, being found in the greatest part of the Brazilian Amazon. T. tricinctus, formerly considered as endemic of the caatinga (Eisenberg and Thorington, 1973; Eisenberg and Redford, 1999; Wetzel, 1985), but also observed in high numbers in some cerrado areas (e.g. Marinho-Filho et al., 1997), had its range basically limited to the semi-arid northeastern region of Brazil.

In the modeling procedure, a relatively high suitability threshold was considered, based on the aims of our study, as described in Section 2.2. Therefore, with less false-positive errors, highly suitable areas were selected (Fielding and Bell, 1997). Moreover, according to the meta-population theory, source populations, which are more abundant and capable of persisting in isolation, mainly inhabit more suitable areas (Andrewartha and Birch, 1954; Hanski, 1991; Harrison, 1991). These source populations are, therefore, of greater conservation value, as they are more able to successfully maintain the meta-population as a whole (Araújo and Williams, 2000; Loiselle et al., 2003). Another pitfall for selecting adequate areas for conservation is the risk of identifying gaps, and selecting priority areas based on a high level of uncertainty that results from the modeling procedures. This issue must be carefully considered in the gap assessment (Loiselle et al., 2003).

For all of these reasons, the distributions of the species, defined by a restrictive and conservative threshold, can appear smaller than they really are, but this procedure is justified by the need of identifying more suitable areas for the persistence of the species.

However, because of a data deficiency observed for the Amazon, our results should be interpreted with care. The resulting models could have left out a significant part of this biome, namely for M. tridactyla and P. maximus, since modeling procedures tend to fit to the input data quite strongly (Phillips et al., 2006). The lack of availability of species list and studies in great part of the Amazon region could have affected the quality of the distributions modeled for this biome. We would thus like to stress the importance of making species lists available whenever possible, as these data are the base for future modeling and meta-analyses.

Results from the gap analysis demonstrated that M. tridactyla and P. maximus benefit greatly from the existence of the SUCU system, since they occur in great part of the Amazon, where the largest IPCU, and most SUCU in the country are located (Fig. 2). If those reserves are actually adequate to preserve native fauna, then...
both species are adequately protected now, and in the future. If we chose to remain conservative, however, we should indicate the urgent need for the expansion of the IPCU system. Nevertheless, we cannot ignore the potential value that sustainably managed reserves can have for the conservation of biodiversity. Efforts must thus be made to ensure that the SUCU system is actually being managed according to its legal objective, and that they maintain significant levels of structurally and functionally biodiverse habitats, even with the presence of human activities (Peres and Zimmerman, 2001). But because many armadillo species, especially the three-banded armadillo (*T. tricinctus*), are subject to an intensive hunting pressure by local communities (Leeuwenberg, 1997; Cullen et al., 2000; Peres, 2000), we stress the fact that the conservation success of these species in indigenous reserves and other protection areas with human occupation must be assessed with care.

Differently from the first two species, the three-banded armadillo (*T. tricinctus*) is far from being adequately protected by either reserve system, which can be explained by the small number of reserves in the caatinga biome (Santos and Tabarelli, 2005). This is now known to be a high endemism region, but historically, it has been neglected by conservation efforts in Brazil (MMA, 2003). We therefore stress the need for greater management actions in the Northeast of Brazil. Also, considering that the central portion of the ranges of *M. tridactyla* and *P. maximus* occur in central Brazil, we also highlight the ecological importance of the cerrado for the conservation of xenathrans. This fact has already been stressed for armadillos (Dasyopodidae family) by Anacleto et al., (2006). More reserves, including those directed at sustainable use, are greatly lacking in both these biomes.

For 2050, modeled distributions illustrate a less pessimistic perspective, compared to predicted effects of climate change on other taxa (Anciães and Peterson, 2006; Marini et al., 2009a; Pounds et al., 1999). It is generally proposed that generalist species are more likely to disperse independently from their interacting species (Araújo and Luoto, 2007; Preston et al., 2008). Although these xenarthran species are specialist insectivores, they opportunistically consume different ant and termite species, which rarely lack in the Neotropical region, and could be expected to behave as expected for generalist species (Anacleto and Marinho-Filho, 2001; Anacleto, 2007; Medri et al., 2003).

Nevertheless, due to the effects of land use change and lack of connectivity among existing reserves, the consequence of climate change on the geographic distribution of many species will probably be closer to that displayed by the null dispersal scenario (Pearson, 2006). Under this less optimistic scenario, all three species are expected to undergo a certain degree of range contraction, as observed for other taxa (Marini et al., 2009a; Peterson et al., 2002b). Nevertheless, they will remain widespread. The occurrence of these species in savanna and semi-arid environments suggest that they are adapted to drier and hotter conditions, which are expected to expand due to climate change in Brazil (Siqueira and Peterson, 2003). Marini et al. (2009a) find that, in the Brazilian cerrado, savanna-dependent bird species are less harshly affected by climate change than forest and grassland-dependent species. This phenomenon may be observed for *M. tridactyla* and

![Figure 4. Total protected areas and the remaining native areas that comprise the future distributions under null dispersal for *Myrmecophaga tridactyla*, *Priodontes maximus*, and *Tolypeutes tricinctus*. The distributions of *M. tridactyla* and *P. maximus* were overlapped, for the indication of priority areas for conservation in regions where they both occur. Priority areas for the Northeast of Brazil are indicated using the distribution of *T. tricinctus* only.](image-url)
P. maximus, as they occur in savanna habitats as well as in forest habitats in the cerrado (Redford and Fonseca, 1986; Redford, 1994).

The final map for the indication of priority areas for future protection indicated conservation opportunities for M. tridactyla and P. maximus in the Pantanal, in the states of Mato Grosso (MT) and Mato Grosso do Sul (MS), and in the northern cerrado region, in the states of Tocantins (TO) and Maranhão (MA). For the conservation of T. tricinctus, opportunities exist in the caatinga of the state of Piauí (PI), in northern Bahia (BA), as well as in great part of the state of Pernambuco (PE), Paraíba (PB), and Ceará (CE), where large tracts of preserved areas still exist (Fig. 4).

5. Conclusion

Even considering the relatively mild effects of climate change on these species, we suggest that attention must be paid to conservation planning that considers future distribution shifts. The strategy we suggest involves paying attention to areas that are suitable today, and will remain suitable in the future are selected for prioritization of planning, as performed in this study (Fig. 4). Identified gaps should be filled in such a way as to remain effective in the face of dynamic threats, such as climate change (Lindenmayer et al., 2010; Gianfrani et al., 2011; Strange et al., 2011), as well as being monitored for assessing its success in the future (Conroy et al., 2011). Also, assessing sites that can maximize connectivity between suitable areas can enhance the possibility of dispersal for a number of species, and maintain the quality of the landscape as a whole (Hannah et al., 2008; Viña et al., 2010). Attention must be paid to areas which still remain standing, especially in biomes with high rates of deforestation, as is the case for the caatinga and the cerrado (Costa et al., 2005; Klink and Machado, 2005; Santos and Tafarelli, 2005).

Moreover, abundance data collected throughout a species’ range, which is lacking for most tropical species, would be important information on how and where climate change is more likely to affect populations’ densities and sizes (Allen et al., 2001; Shoo et al., 2005). Finally, the identification of gaps must serve as incentive for the urgent planning of new conservation strategies, taking other taxa into consideration (Guisan and Thuiller, 2005; Marini et al., 2009b).

Acknowledgements

We would like to thank the Earthwatch Institute, the Monsanto Foundation, the Memorial Zoo and the CI Brasil, for their financial support for the Jaguar Conservation Fund (JFC) projects; the landowners for giving permission for the JFC crew to work at their properties; the IBAMA for granting research licenses for work in protected areas; and the many volunteers and trainees who helped in the field. This study also received a technical support from the National Forest, Rondonia state. Biota Neotrop. 8, 231–234.


