Deforestation and biodiversity conservation in Mexico

Víctor Sánchez-Cordero[1], Patricia Illoldi-Rangel[1,4], Tania Escalante[2], Fernanda Figueroa[1], Gerardo Rodríguez[3], Miguel Linaje[1], Trevon Fuller[4], and Sahotra Sarkar[4]

[1] Departamento de Zoología, Instituto de Biología, Universidad Nacional Autónoma de México. Apartado Postal 70-153, México, D.F. México 04510.

[2] Museo de Zoología "Alfonso L. Herrera", Departamento de Biología Evolutiva, Facultad de Ciencias, Universidad Nacional Autónoma de México. Apartado Postal 70-399, México, D. F. México 04510

[3]Laboratorio de Macroecología, Departamento de Ecología de la Biodiversidad, Instituto de Ecología, Universidad Nacional Autónoma de México. Apartado Postal 70-275, México, D.F. México 04510

[4] Biodiversity and Biocultural Conservation Laboratory, Section of Integrative Biology, University of Texas at Austin, Austin, TX 78712-1180, USA.

Abstract

Deforestation is one of the main factors negatively affecting the conservation of biological diversity. We assess the impact of deforestation on biodiversity in Mexico by quantifying its effect on (1) mammal species' distributions, (2) delineation of biogeographical regionalization, (3) the effectiveness of conservation area networks to prevent biologically deleterious land use/land cover change, and (4) area prioritization for biodiversity conservation. Deforestation has a significant impact on species' distributions with habitat loss ranging from 10-90%, leading in some cases to high risks of extinction. Significant changes also occur in the delineation of biogeographical provinces (bioregionalization) due to deforestation because several areas with unique fauna show high rates of natural habitat reduction, leading to potential ecological and biogeographical changes in species interactions and distributions. A variety of decreed natural protected areas now show increasing pressures due to potential land use conversion from natural

habitat to agriculture and urban settlements which threatens their biodiversity content. Area prioritization for biodiversity conservation is seriously negatively affected by deforestation because a significantly increased total area is required to conserve endemic mammals as compared to what would have been adequate 30 years ago before significant loss of forest cover. These trends urge immediate collaborative actions between academic, governmental, and NGO sectors to prevent and reduce further deforestation nationwide.

Introduction

High rates of deforestation threaten biodiversity conservation worldwide (Dale *et al.*, 1994, Lidlaw, 2000; Sala *et al.*, 2000; Kinnard *et al.*, 2003), causing land degradation (Riezebos and Loerts, 1998; Islam and Weil, 2000), local, regional, and global climatic changes (Houghton *et al*., 1999; Chase *et al.*, 2000), and loss of ecosystem services (Vitousek *et al.*, 1997). Of major concern are impacts of deforestation on individual species' distributions and, consequently, their conservation status (Wilson, 1988; Reaka-Kudla *et al*., 1997; Mace *et al*., 1998; Sánchez-Cordero *et al.*, 2005), as well as on natural protected areas (NPAs) which form the foundation of most strategies of environmental conservation. Although international forums have proposed that the global network of Natural Protected Areas (NPAs) should reach 10% of the Earth's surface (IUCN 2005), conservation of this area is not guaranteed because many NPA networks face a variety of threats and suffer from a variety of degradation processes worldwide (Carey *et al.* 2000).

One approach for evaluating the impact of deforestation on biodiversity conservation is to quantify land use-land cover changes (LULCC) in dominant vegetation types because of habitat loss and the accompanying loss of biodiversity. Several studies have identified vegetation types with high species richness and endemicity suffering considerable deforestation; such hotspots occur primarily in tropical and montane regions (Wilson, 1988; Laurance and Bierregaard, 1997; Mittermeier *et al*., 1998; Myers, 1998; Kinnaird *et al*., 2003; Rodrigues *et al*., 2004), especially in tropical rainforests, of which over 70% has been deforested (Myers, 1998; Laurance and Bierregaard, 1997; Heaney *et al*., 1999).

Mexico is a megadiverse country showing high rates of deforestation posing threats to its biological conservation. Annual rates of deforestation are over 1.2% (Masera *et al*., 1997; Arriaga *et al*., 2001; FAO, 2001; Mas *et al*., 2004). Further, an estimated 90% of the original humid tropical forest has been converted into agrosystems or urban settlements, presumably resulting in significant biodiversity loss (Toledo *et al*., 1989; Mittermeier *et al*., 1998; Dirzo and García, 1992; Arriaga *et al*., 2001).

Recent developments in ecological niche modelling provide the possibility of insight into these challenges (Soberón & Peterson, 2005); by integrating known occurrences with digital maps of relevant environmental parameters, areas suitable for species can be characterized and projected as potential geographic distributions (Soberón & Peterson, 2005; Soberón, 2007). Terrestrial mammals have been extensively used as a model group in different ecological, biogeographical and conservation studies. Some of the reasons for this are that there exists an extensive tradition of their study, their individual distributional areas are more or less well-known, and their phylogenetic relations are typically well-understood. Mammals in México are well represented in biological collections: there are 28 national collections which compile 162,288 specimens (Espinoza *et al*., 2006), and there are collections in Canada and the USA in which there exist approximately 242,420 specimens (López-Wilchis, 2006). Also, for the majority of biological collections, specimen data are accessible online (e. g., http://manisnet.org/, http://unibio.ibiologia.unam.mx/ http://www.gbif.org/, http://www.conabio.gob.mx/remib/doctos/remib_esp.html), which facilitates their use and analysis.

We address diverse negative impacts of deforestation on biodiversity conservation in Mexico using terrestrial mammals as a case study. Specifically, we (1) analyze changes in estimated range loss due to deforestation by determining the remnant untransformed habitats within species' potential distributions based on ecological niche modeling, and identify regions of potentially high species' endangerment and extinction risk; (2) delineate the impact of deforestation on biogeographic region delineation based on patterns of endemism by contrasting species' potential and extant distributions; (3) evaluate the extent of change of areas in which the original vegetation was completely transformed to human settlements, and agricultural and grazing lands in federal NPAs, as a measure of the impact of deforestation on conservation policies; and (4) carry out area prioritization for biodiversity conservation targeted to conserve endemic mammals, comparing what is required now to what would have been adequate 30 years ago (before significant loss of forest cover).

Deforestation and biodiversity loss

Many previous studies have linked deforestation of predominant vegetation types with biodiversity loss (Dirzo and García, 1992; Laurance and Bierregaard, 1997; Heaney *et al*., 1999). Although these studies are relevant insofar as they correlate general trends of deforestation and biodiversity loss, these approaches were limited insofar that they did not assess the impact of deforestation on individual species. Recently developed methodologies for modeling species' distributions are based on the modeling species' ecological niches (*sensu* Grinnell [1917] – the suite of environmental conditions within which a species can maintain its population without immigration (MacArthur, 1972). These methodologies use occurrence data, environmental layers, and a GIS platform to predict species' distributions. Coarse-grained ecological niche models can identify geographic areas potentially suitable for each species (Peterson *et al*., 1999; Stockwell and Peters, 1999; Peterson, 2001; Anderson *et al*., 2003). We believe that such geographical predictions of niches can provide a framework for understanding how habitat loss impacts distribution patterns of individual species and hence, their conservation status (Sánchez-Cordero *et al*., 2001, 2004).

We modeled ecological niches for the continental endemic Mexican mammals, and estimated range loss due to deforestation using the 2000 land use and vegetation map by determining the remnant untransformed habitats within species' potential distributions. We only included endemics to ensure that the modeled ecological niches projected as species' potential distributions remained within México (for which we had all the relevant data). To overcome geographic biases in delimiting species' distributions, we generated models of ecological niche for each species, using the Genetic Algorithm for Rule-set Prediction (GARP; Stockwell and Peters 1999; available for download at www.lifemapper.org/desktopgarp), which has proven to be quite robust for predicting mammal distributions in México (Illoldi *et al*., 2004). Our main results indicate that more than one-fourth of the endemics (23 out of 85 species) have lost >50% of untransformed habitat within their potential distributions. Two endemics showed a particularly drastic distributional loss of >90%, and another two showed losses of >80%; only 10 endemics retained >80% of untransformed habitat within their distributional areas (Sánchez-Cordero *et al.*, 2005). Overall, 35% of untransformed habitat has been lost nationwide, with major losses occurring mostly in tropical habitats. Most endemics (61 of 85 species, 72%) showed a higher proportion of transformed habitat within their potential distributions compared to the overall proportion of transformed habitat at the national level. Endemics showing significant areal reductions with low remnant untransformed habitat coverage within their potential distribution can be interpreted as posing an extinction threat. Nearly all endemics with large ranges lost a large proportion of their potential habitats, whereas species with small ranges showed both high and low loss, indicating that a broad distribution does not preclude a high extinction risk. Endemic species can be particularly vulnerable to extirpation in certain parts of their ranges, in which only small fragments of untransformed habitat remain.

No significant correlation existed between the proportion of untransformed area and the original distributional area ($r = 0.07$, $p > 0.1$); rather, the proportional area of untransformed habitat appeared more dependent on geographic location. For example, 25 out of 37 (68%) endemic species retaining <60% of untransformed habitat in their distributions occurred in eastern Mexico and central Mexico. Conversely, 45 out of 48 (94%) endemic species retaining >60% of untransformed habitat in their potential distributions occurred in other regions (Sánchez-Cordero *et al.*, 2005). Historically, the eastern and central regions in Mexico have seen deforestation and urbanization increases since the 1960s, converting 70% of untransformed habitat into agrosystems and rural or urban settlements (Toledo *et al.*, 1989; Challenger, 1998; Arriaga *et al.*, 2001) (Fig. 1). Extinction risk for endemics in these regions appears higher compared with other regions of the country. Nearly 40% of Mexican endemics are restricted to these regions (Hall, 1981; Ceballos *et al.*, 1998; Sánchez-Cordero *et al.*, 2005). If habitat loss across the landscape is assumed to occur at random, we expect more variability in the loss rates of potential habitat for species with small rather than large distributional ranges. Interestingly, most endemics restricted to these regions have small distributions (Hall, 1981), showing high variability in their proportion of untransformed distribution (Sánchez-Cordero *et al*., 2005). These results suggest that populations face different extirpation risks depending on geographic location, thus providing a geographic context for designing regional and local conservation plans. This criterion can be extrapolated generally to other faunistic groups as a useful tool for defining conservation priorities (Ceballos and Rodríguez, 1993; Mace *et al*., 1998; Hilton-Taylor, 2000; Kinnaird *et al*., 2003; Sánchez-Cordero *et al*., 2004, 2005; Fuller *et al*., 2006). However, while habitat loss is recognized as a critical factor for extinction risk, other factors, such as hunting pressure, and species' biological properties are also important when considering conservation status (Laurance *et al.* 2006; see also www.conabio.gob.mx and www.ine.gob.mx).

Our distribution models are based on the assumption that coarse-scale conversion of natural habitats into agrosystems or human habitation (rural or urban settlements) results in non-viable conditions for species (Egbert *et al*., 1999; Peterson *et al*., 2000; Ortega-Huerta and Peterson, 2004; Sánchez-Cordero *et al*., 2004). Ecological support for this idea comes from the hypothesis of general niche conservatism which has been tested for a diverse range of fauna in México and elsewhere (Peterson *et al*., 1999; Peterson and Vieglais, 2001; Peterson and Holt, 2003). Theoretically, rapid adaptation to new environments produced by anthropogenic habitat transformation is unlikely, and populations may not be able to persist without significant immigration from adjacent natural habitats (Peterson and Holt, 2003). This effect reinforces claims of the importance of conserving untransformed habitats. These factors are likely to hold for locally-adapted endemic species.

Our model does not specifically predict which endemics may be tolerant to transformed habitat, as is known to be the case for some mammals (see Fisher *et al*., 2003; Sánchez-Cordero and Zepeda, 2003; Isaac and Cowlishaw, 2004). Moreover, because we used a coarse forest pixel resolution, the areas assumed to be suitable habitat may include portions of transformed habitat. This would lead to an underestimation of the loss for species' potential distributions. Other factors not included in our model may also contribute to population extirpation, such as illegal hunting, and biological attributes of species, home range, fecundity rate, and rarity in fragmented habitats (Mace *et al*., 1998).

These methodological limitations do not affect our main conclusions based on the observed patterns of reductions in species suitable niches due to anthropogenic habitat transformation. From the perspective of biodiversity conservation, our approach satisfies the precautionary principle: even where it requires subsequent refinement and modification, it is conservative about what constitutes viable habitat for species; as such, it will have done no harm to conservation aims (Sarakinos *et al*., 2001; Cooney & Dickson 2005). If some areas are selected for conservation, but other areas also allow the persistence of species, the use of our model will not result in detriment of potential suitable habitat for conserving species (Sarkar, 2004). We propose to expand our analyses more broadly to other taxa in order to identify areas of potential general extinction risk, in which conservation investment or habitat restoration would have a strong impact in preventing biodiversity loss. A clear result of these analyses is that a regional, rather than taxonomic, focus should guide mammal conservation assessment and planning in México (Sánchez-Cordero *et al*., 2004, 2005; Fuller et al., 2006, 2007).

On a broader scale, our approach can be incorporated into current methodologies for assigning species' global conservation status (Hilton-Taylor, 2000), providing quantitative estimates of likely extent of the loss of the geographic distributions of individual species. Increasing efforts supported by universities and governmental agencies are leading to compilation of large databases of georeferenced species point occurrences. These developments will facilitate conducting analyses similar to ours for a wide range of biotic groups anywhere in the world.

Deforestation and biogeographical regionalization

Deforestation affecting species' distributions has an impact on biogeographic region delineation, because deforestation induces changes in historical patterns of species' distributions (Escalante *et al*., 2007a,b). Biogeographic regionalization consists of delineating a hierarchy of geographic areas derived from successively nested areas of endemism, in which the province is the smallest area of endemism to be identified, followed by dominions, regions, and realms. Because areas of endemism are delineated using individual species' distributions, we have good reason to believe that distributional patterns derived from individual distributions change with deforestation.

Hierarchical biogeographic regionalizations are inferred from historical patterns of endemism but these may be modified by transformation of natural habitats. We analyzed the impact of deforestation on patterns of endemism by contrasting potential and extant distributions using terrestrial mammals in Mexico as a case study. Specifically, we showed changes in distributional patterns of terrestrial mammal under two scenarios: by modelling species' ecological niches projected as potential distributions using the original natural vegetation map (t_1), and projected as extant distributions using the 2000 land use and vegetation map (t_2). We generated models of ecological niche for 429 species, using GARP (see above) and built two matrices to develop the regionalization: matrix-t_1, using species' distributions t_1, and matrix-t_2, using species' distributions t_2. Both matrices were used in a Parsimony Analysis of Endemicity (PAE; Morrone, 1994) for 248 quadrats nationwide. For t_1 and t_2, characteristic and endemic species were mapped and these species and the cladograms obtained were used for biogeographic regionalizations.

We observed significant changes between the t_1 and t_2 cladograms resulting in dramatic changes in biogeographical patterns at major scales. We found an obvious delineation of regions in t_1, but not in t_2, suggesting a change in historical distributional patterns by which the boundary between Nearctic and Neotropical regions virtually disappears; thus such biogeographic transitional zones which were formed in historical times have suffered dramatic changes as a consequence of deforestation. Surprisingly, the Sierra Madre Occidental province was not identified under the t_2 scenario (Fig. 2). Species may respond to deforestation by (1) reducing their distributional area only to remnant suitable habitats, (2) modifying their distributional area using perturbed habitats, and/or (3) moving to other remnant suitable habitats (Holt, 1990). If each species responds in different ways to drastic ecological changes, then shared distributional patterns are modified and, in that sense, lost.

Changes in biogeographic regionalizations should be expected over ecological or evolutionary times, produced by unique geologic, geographic, and climatic changes at a global scale. We showed that changes in species' distributions attributable to deforestation can result in significant changes in biogeographical regionalization using terrestrial mammals as a case study. If further deforestation continues, it is likely that additional regions holding unique patterns of species' distributions may disappear with unknown consequences for ecological-evolutionary processes (Wilson, 1988). Similarly, deforestation and climate change can also act in concert to produce rapid changes in species' distributions thus affecting biogeographic regionalization as well (Peterson *et al*., 2002). Studies incorporating biogeographic regionalization of areas of endemism and transition zones can help identify conservation priorities.

Deforestation and Natural Protected Areas

Networks of Natural Protected Areas (NPAs) constitute the foundation of policies for maintaining biological diversity, the livelihood of local communities, and the provision of ecosystems services (Ervin, 2003; IUCN 2005). Notwithstanding, NPAs face several threats worldwide, such as deforestation and other land use-land cover changes (LULCC) such as other forms of land degradation, exotic species invasions, illegal logging, hunting and extraction of resources, among others (Carey *et al.*, 2000). Recent trends of high natural landscape transformation due to human activity jeopardize the integrity and health of ecological systems, posing a threat to NPAs which frequently consist of the only remaining natural habitat of many species.

Deforestation poses a critical threat to biodiversity conservation in megadiverse México, and this challenge is further exacerbated by the existence of a growing and highly dispersed rural population increasingly demanding direct use of natural resources (Sarukhán *et al*., 1996). Conservation strategies in Mexico rely heavily on NPAs. In 2003, there were 160 officially decreed NPAs to protect terrestrial and marine ecosystems, varying in size and management category, and covering approximately 9% of the nation (CONANP, 2003; http://www.conanp.gob.mx/anp/anp.php). As in most developing countries, NPAs in Mexico have been historically inhabited by local communities, many of which have been living in these areas for centuries. An estimated 1,404,516 people were living in 4,485 localities inside NPAs in 2000 (CONANP, 2003). Many factors influence the NPAs' degradation by human activities including demographic dynamics, local and non-local economic interests and activities, poverty, market demand and prices of local products, governmental policies, and conflicts among different social groups over the control and use of natural resources (Barbier and Burgess, 1996; Ghimire and Pimbert, 1997; Barbier and Burgess, 2001; Lambin *et al.*, 2001; Geist and Lambin, 2002). Consequently, there is an increasing concern over the NPAs' capacity to fulfill their conservation goals (Hockings, 1998; 2003).

An evaluation of degradation processes within NPAs is essential for conservation planning, especially for detecting areas of particular concern, and for designing strategies to prevent biodiversity loss (Sánchez-Cordero and Figueroa, 2007; Sánchez-Cordero *et al.*, 2007). We evaluated land use - land cover change (LULCC) processes in Mexican NPAs from 1993 to 2002, assuming a correlation between natural vegetation cover reduction and the loss of biological diversity and the ability to provide ecosystem services. We used the extent and rate of change of highly transformed areas inside NPAs as a surrogate of LULCC. These included areas transformed due to agriculture, induced and cultivated pastures, forestry plantations, and human settlements. We estimated the extent and rate of change of transformed areas using land use and vegetation maps produced by the Instituto Nacional de Estadística, Geografía e Informática for 1993 and 2002 (INEGI, 1993; 2005), the map of the decreed federal NPAs of Mexico (CONANP, 2003), and a GIS platform (Figs. 3 and 4). We estimated LULCC rate as the annual percentage of change of transformed areas relative to the total evaluated area, as follows:

$$LULCCR = \frac{S_2 - S_1}{S_t} \times \frac{100}{N}$$

where $LULCCR$ = change rate, S_1 = initial transformed surface area, S_2 = final transformed area, S_t = total evaluated area, and N = time difference in years.

The extent of transformed area was estimated for 89 terrestrial federal NPAs decreed before 2003, and larger than 1,000 ha (CONANP, 2003); the rate of change was estimated for 69 of these areas decreed before 1997. The main results showed that approximately one-fourth of Mexico had been converted to highly transformed areas by 2002 (Fig. 2) reflecting a history of inadequate land use and management policies for the conservation of natural resources. Most NPAs suffered less habitat transformation (mean of 16% of their area), showing smaller percentages of transformed area than nationwide. Fifty percent of NPAs showed less than 10% of transformed area, but one third showed more than 30% of transformed area; 8% showed more than 50% of transformed area.

These trends indicate high threats to biodiversity conservation in these NPAs and should be viewed with concern.

The growth rate of transformed areas nationwide was 0.2% annually; this rate means that roughly 2% of Mexico is anthropogenically transformed each decade. The mean annual rate of change in NPAs was of 0.03%, with more than half of NPAs (57%) showing a reduction of transformed areas (rates ranging 0 to -1.5); in these NPAs, natural vegetation is either not being reduced spatially, or is recovering. Somewhat less than half (43%) of NPAs showed increments in transformed area (Fig. 5), although with lower rates than the national level. However, as many as 20% of NPAs showed rates of change higher than nationwide.

We expected lower estimates for transformed areas and rates of LULCC in NPAs. The lower rate change in NPAs may result from the combination of environmental policies and resources, along with the fact that many of these areas show conditions that result in lower suitability for economic activities, such as greater isolation due to less access to roads, or because soil types and topographic characteristics compromise potential agricultural productivity (Pressey *et al.*, 2002, Brandon *et al.*, 2005, Mas, 2005). This trend is expected in a country such as México because of its long history of intensive human occupation, because of which the best suitable areas for agricultural production have been under use for a long time with the consequence that decreed NPAs are usually located in areas that are less suitable for such economic activities (Cantú *et al.*, 2004; Sánchez-Cordero and Figueroa, 2007; Sánchez-Cordero *et al*., 2007).

Most NPAs showed a reduction or slight increment in transformed areas, but many had processes of LULCC that nevertheless constitute a threat to conservation. The design and implementation of strategies of vegetation recovery such as ecological restoration in areas of particular concern is urgently needed, along with research focusing on causal factors underlying LULCC processes. Policies aimed at sound economic activities based on the sustainable use of natural resources by local people should be supported by government agencies, both in communities living inside and surrounding NPAs. This is particularly important, as many of these rural communities show conditions of high marginalization (Nadal, 2003). Future work should link spatial overlapping of priority areas for conservation and biogeographic regionalization with location of NPAs in which transformed areas are both decreasing and increasing. Such analyses are important inputs for systematic conservation planning of the biological diversity in Mexico.

Deforestation and the cost of postponing biodiversity conservation

As in many other developing countries, land conversion in México during the last thirty years has been extensive. Tropical and temperate forests in México are disappearing at high annual rates (Masera *et al*., 1997; Mendoza *et al*., 1999) with increasing agricultural lands, shrubs, and pastures for cattle (Bocco *et al*., 2001). These reductions of natural habitat can extirpate many vertebrates such as mammals from regions severely affected by deforestation (Peterson *et al*., 2006). Conversion to agricultural use creates habitat that is typically unsuitable for at-risk mammal species (Ceballos *et al*., 2005; Sánchez-Cordero *et al*., 2005; Stoleson *et al*., 2005), which is particularly critical, as Mexico's mammal fauna ranks second in richness worldwide, and of which 30% is endemic (Fa and Morales, 1993). Moreover, Mexican endemic mammals are of special conservation concern because they are underrepresented in international treaties about at-risk species (Ceballos *et al*., 2002).

Recently, a database with remote-sensed data was created for the extent and rate of LULCC in Mexico, which includes nationwide land use and vegetation maps since the 1970s (Sorani and Alvarez, 1996; Mas *et al*., 2004; see www.igeograf.unam.mx, and www.conabio.gob.mx). However, such data do not indicate how land conversion affects strategies for the conservation of biodiversity (Kinnaird *et al*., 2003). In particular, the effects of land use change on the required size of adequate biodiversity conservation area networks were largely unknown. We combined the database on land conversion with ecological niche modeling of 86 Mexican endemic mammals projected as species' distributions, and (1) analyzed distributional shifts in the recent past by quantifying the impact of land use patterns on species' distributions from 1970 to 2000, and (2) assessed how shifts in species' ranges affect the design of optimal conservation area networks. We hypothesized that deforestation resulting in reduction and fragmentation of natural habitats has a cost in the amount of land that must be placed under protection to represent mammalian biodiversity today. We predicted that a greater area of land would be required today compared to the area that would have been required to protect endemic mammals at equivalent levels thirty years ago using the same target level of representation of the species in both cases (Fuller *et al*., 2007).

Our main results verify this prediction and indicate that significantly more land is required to represent mammal habitat in 2000 than when sites are selected with the 1970 distribution (41 *vs* 89 %, respectively), and > 60 % of the optimal conservation areas that would have been selected in 1970 had been transformed by 2000. Moreover, for 90% of the endemic mammals, the potential distributions in 2000 comprised fewer sites than the potential distribution in 1970. If a conservation area network had been implemented even as late as 1970, mammal habitat could have been protected in a considerably smaller set of sites than what is required today owing to land degradation and concomitant thinning of species' ranges. Consequently, area prioritization for biodiversity conservation is negatively affected by deforestation because a significantly increased area is required to conserve these endemics compared to what would have been adequate 30 years ago before significant loss of forest cover.

Conclusions

In sum, deforestation has dramatic negative effects on species' distributions, biogeographic regionalization, the performance of NPAs, and the design of conservation area networks. All these factors emphasize the threat to biodiversity conservation posed by deforestation. These trends urge immediate collaborative actions involving academic, governmental, and NGO sectors to prevent and reduce further deforestation nationwide. These conclusions are hardly new but the methods we have developed allowed us to quantify all these effects. This quantification required the systematic use of remote-sensed data, GIS platforms, and a variety of algorithms and other computer-aided techniques. Moreover, our analyses can be linked with recent theoretical framework in conservation as systematic conservation planning (Margules and Sarkar, 2007) to facilitate implementation at regional and local scales requiring immediate conservation actions. Though our analyses were all performed in the context of México, the methodologies developed were fully general and can be exported unchanged to any other (terrestrial) context. Other biodiversity hotspots that are experiencing deforestation at a rate at least as high as that of Mexico include Burma, Ecuador, Indonesia, the Phillipines, and Sri Lanka (Myers *et al*., 2000; United Nations Food and Agriculture Association, 2007). The findings presented here on the cost of postponing biodiversity conservation in Mexico may also apply to those other hotspots. Testing this hypothesis will require the construction of a multidate databases on LULCC, which are not presently available for most developing countries.

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Figure legends

Figure 1. Effect of land conversion on endemic mammal distributions in Mexico. Black areas include the habitat of the Mexican agouti (*Dasyprocta mexicana*), a charismatic and economically important mammal typically inhabiting rainforest. The modeled distribution of *D. mexicana* showed a reduction of 33.5% from 1970 (A) to 2000 (B).

Figure 2. Area cladogram showing the final biogeographic regionalization based on extant mammals distributions in Mexico. Deforestation significantly reduced the distribution of many mammals which affected historical biogeographic region delineation. See text for further details.

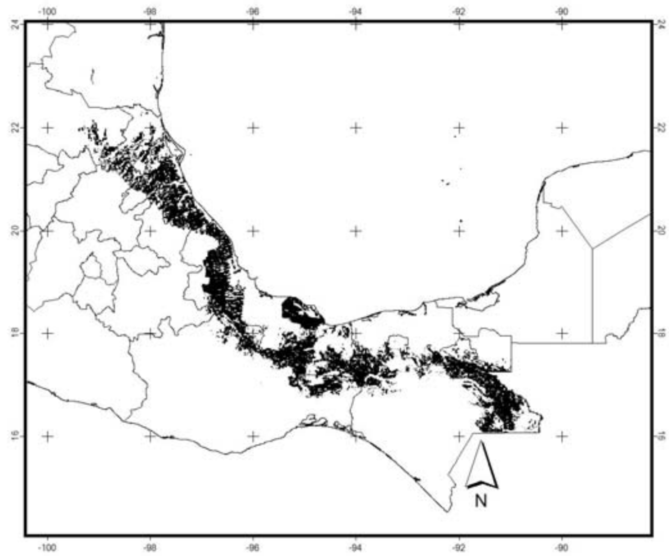
Figure 3. Map depicting the decreed terrestrial federal Natural Protected Areas of Mexico (CONANP 2003), covering approximately 9% of the nation (see www.conanp.gob.mx).

Figure 4. The 2002 transformed areas of Mexico including agriculture fields, induced and cultivated pastures, and urban settlements (black area). Large portions of deforested regions occur predominantly in central, eastern and south of Mexico (see www.inegi.gob.mx).

Figure 5. (A) Range (%) of transformed areas in 89 NPAs in Mexico; national mean value is 26%. (B) Annual rate of change of transformed area in 69 NPAs in Mexico; national mean value is 0.2%.

Figure 6. Optimal conservation areas for Mexican endemic mammals in 1970 and 2000 obtained by including 10% of each species' potential distribution. The optimization was initialized with the natural protected areas. Note the large amount of additional conservation area required to conserve the same number of endemics in 2000 (dark gray) compared to 1970 (light gray). Taken from Fuller *et al*. (2007).

A



B

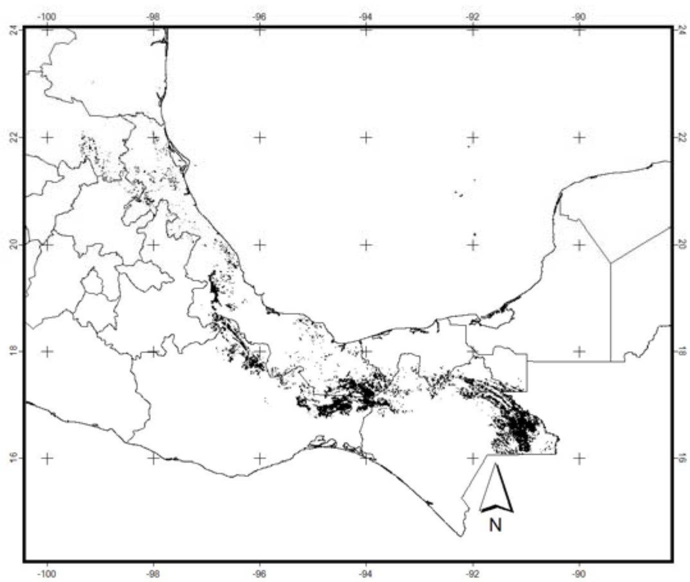


Figure 1.

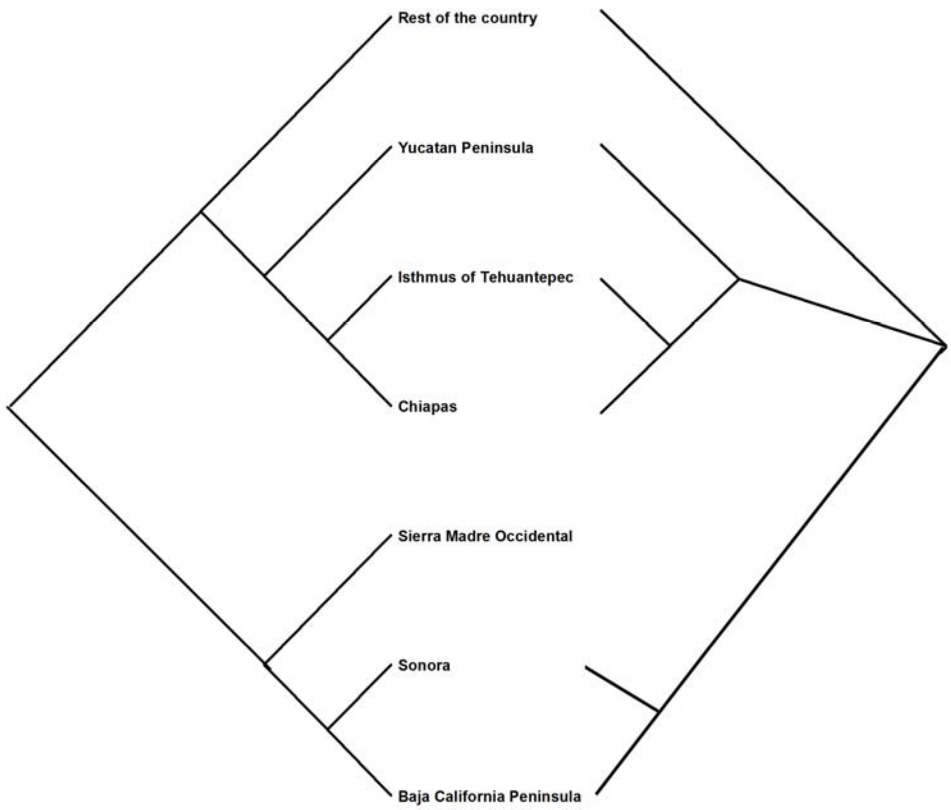
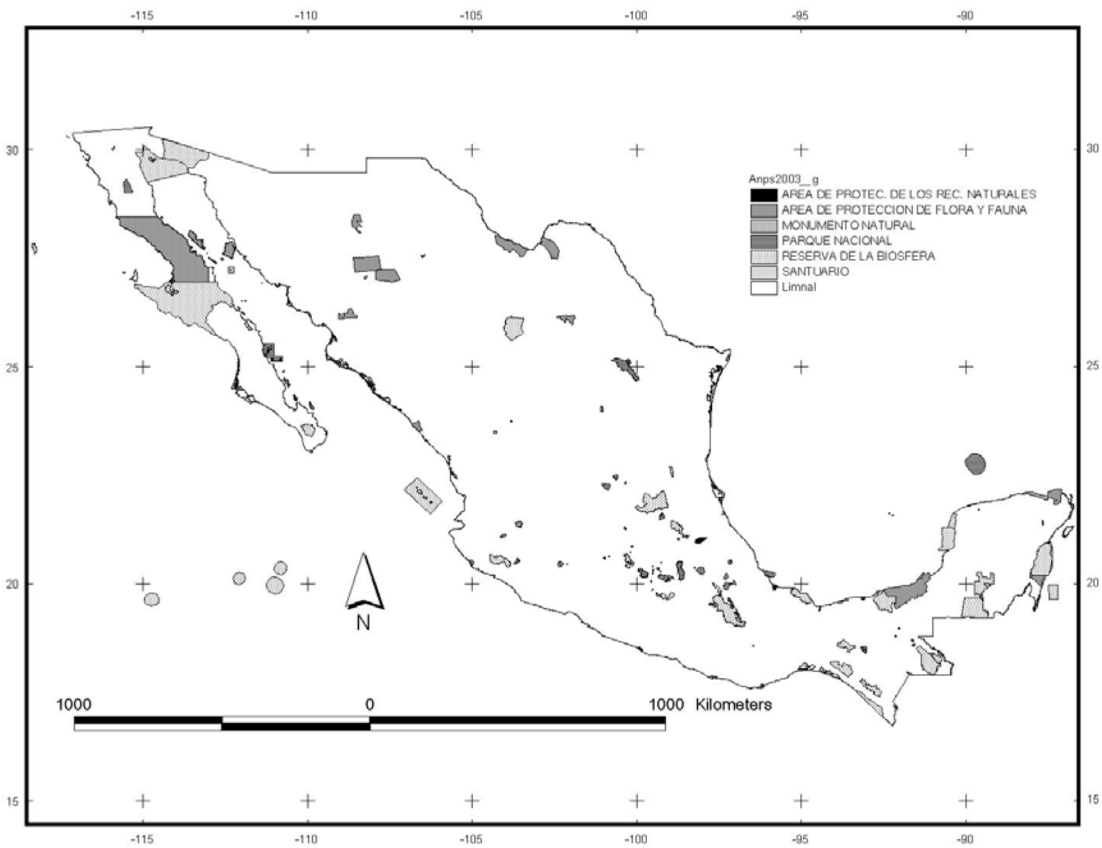

Rest of the country
Yucatan Peninsula
Isthmus of Tehuantepec
Chiapas
Sierra Madre Occidental
Sonora
Baja California Peninsula

Figure 2.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AREA DE PROTEC. DE LOS REC. NATURALES
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MONUMENTO NATURAL
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-95
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30
25
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Figure 3

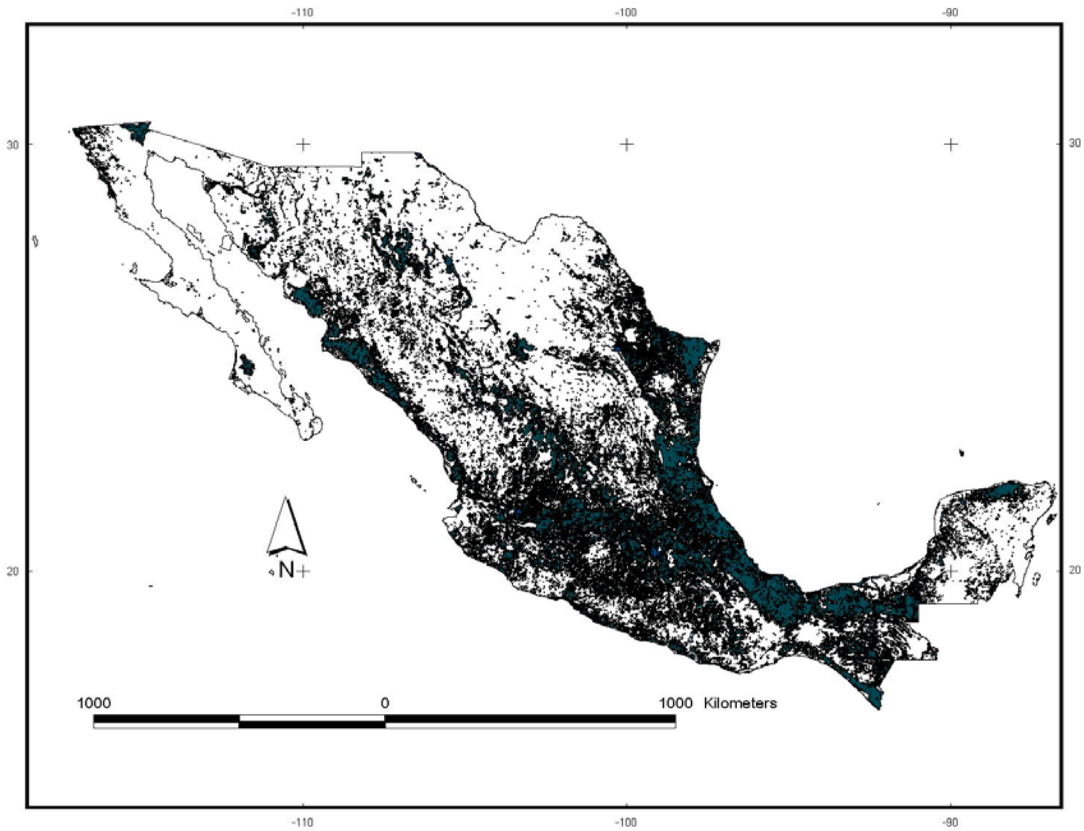
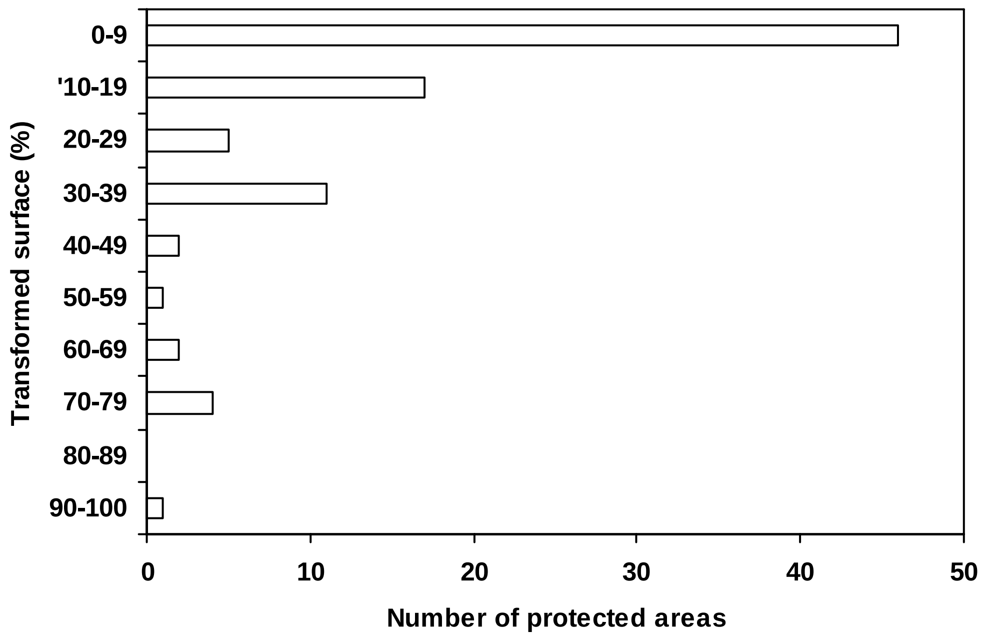


Figure 4.

A


Transformed surface (%)
0-9
'10-19
20-29
30-39
40-49
50-59
60-69
70-79
80-89
90-100
0
10
20
30
40
50
Number of protected areas

B

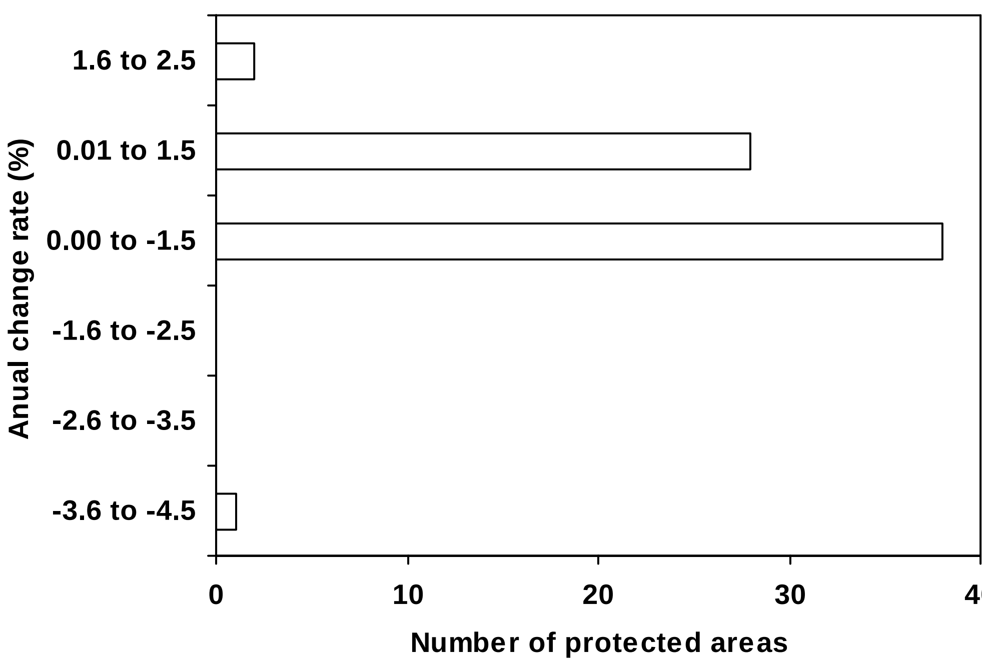

Anual change rate (%)
1.6 to 2.5
0.01 to 1.5
0.00 to -1.5
-1.6 to -2.5
-2.6 to -3.5
-3.6 to -4.5
0
10
20
30
40
Number of protected areas

Figure 5.

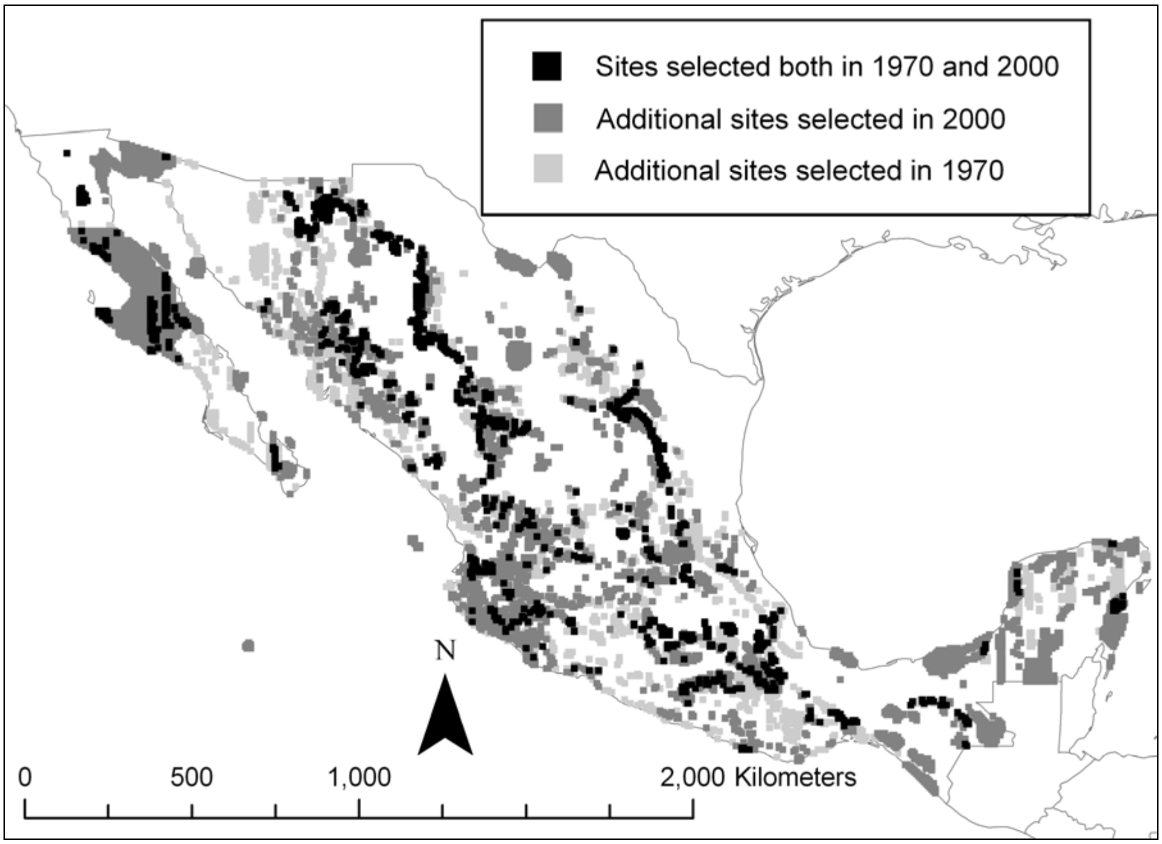

Sites selected both in 1970 and 2000
Additional sites selected in 2000
Additional sites selected in 1970
N
0
500
1,000
2,000 Kilometers

Figure 6.