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Reconciling Biodiversity Conservation, People, Protected Areas, and Agricultural Suitability in Mexico

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Summary. — Methods are needed to identify priority areas for biodiversity conservation that minimize conflict with agricultural productivity. Analysis of georeferenced datasets for breeding birds, mammals, and amphibians in Mexico indicates that only 94 of 3,040 areas are needed to include all unprotected species within a reserve system. An examination of socioeconomic data reveals that in most of these 94 areas, opportunities exist to develop reserve networks that conserve biodiversity without adversely affecting existing human settlement, land use, or agricultural productivity. Planning that simultaneously considers infrastructure development, agricultural suitability, and protected areas can conserve biodiversity, increase agricultural production, and support rural livelihoods.

Key words — Latin America, Mexico, biodiversity conservation, agricultural productivity, protected areas, locational modeling

1. INTRODUCTION

As we enter the 21st century, much of the world's remaining biological diversity is in danger of disappearing (Pimm *et al.*, 2001). Globally, no less than 25% of all mammals, 12% of birds, and 20–30% of reptiles and amphibians are threatened (Hilton-Taylor, 2000). Pressure to convert an additional one-third of global land area from natural habitat within the next 100 years will lead to competition among rural land uses (United Nations, 1999; World Resources Institute, 2000). Amidst this pressure, improved management of existing protected areas (reserves) and the creation of new reserves are essential to avoid major species losses. Reserves maintain the array of environmental services upon which humans depend while maintaining biodiversity (Balmford *et al.*, 2002; Balvanera *et al.*, 2001). Loss of ecosystem services may disproportionately affect the poor, since they lack the means to compensate or replace these services in other ways. However, creating reserves to stop or reduce land conversion and extractive uses can also adversely affect rural residents (Brandon, Redford, & Sanderson, 1998; Bruner, Gullison, Rice, & Fonseca, 2001; Sánchez-Azofeifa, Daily, Pfaff, & Busch, 2003; Wells & Brandon, 1992). [1]

Delegates to the most recent (2003) World Parks Congress in Durban, South Africa, generally agreed that although the global reserve system must be expanded to avoid plant and animal extinctions, reserve benefits and costs are not equally distributed. Global benefits are often greater than local ones, and within countries, reserves may impose higher opportunity costs on the poor, although in some cases, the benefits that the poor derive from these

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areas exceed the limitations imposed on local livelihoods. The designation of new reserves that halt habitat conversion and species extinction must be based on sound information on the ecological, socioeconomic, institutional, and financial context (Brandon *et al.*, 1998; Cowling & Pressey, 2003). Building on such information, it is possible to design reserves that are integrated into the landscape and that support, rather than detract from, local lives and livelihoods.

The following paper describes a method for conservation planning that considers both biodiversity protection and agricultural suitability using data on Mexico. The study demonstrates that it is possible to include all species of breeding birds, mammals, and amphibians that remain unprotected by the existing reserve system into a relatively small number of reserves with low agricultural productivity. In both forested and unforested settings, such methodologies demonstrate that focused, strategic planning for conservation and development can maintain biodiversity and ecosystem services without imposing serious restrictions on agricultural production.

2. THE CONTEXT OF CONSERVATION IN MEXICO

Mexico is a priority region for global conservation, ranking among the top five countries of the world for endemism [2] of both vascular plants and vertebrate species (Mittermeier, Myers, Thomsen, da Fonseca, & Olivieri, 1998). Approximately 31–33% of mammals, 60–62% of amphibians, 49% of freshwater fish, and 40–50% of species of flowering plants occurring in Mexico are endemic (US Agency for International Development, 2002). This high endemism is related to the wide-ranging topography, numerous climatic zones, and Mexico's location bridging tropical and temperate zones. Despite more than 10,000 years of human habitation, rich biodiversity has persisted in Mexico into the 21st century. However, Mexico's biodiversity is highly threatened due to forest fragmentation, natural habitat loss and degradation, pollution, unsustainable and illegal land and resource use, collection and trade of plants and wildlife, and global climate change (US Agency for International Development, 2002).

Mexico's population is 25% rural, with this subsection of society being highly dependent on agriculture. Poverty (24% of the total population living under US$2 per day) is skewed toward certain regions and coincides with land scarcity, often leading to colonization of intact forest, including both protected areas and lands recognized as belonging to indigenous peoples (Human Development Index, 2003). Forested areas contain approximately 11% of Mexico's population (10 million people). More than 8,400 rural communities own forest resources (Segura, 2000) and Mexico contains a large number of communal forest enterprises (Bray *et al.*, 2003). Unlike most other countries in the world, 80% of forests are owned at the community level (Klooster & Masera, 2000; Segura, 2000). However, forestry represents the major economic activity in only 421 communities concentrated in two states. Despite years of deforestation, a considerable amount of forest area remains in Mexico, and annual deforestation rates over the final quarter of the 20th century were less than 0.5% (Velázquez, Mas, Palacio, & Bocco, 2002; Velázquez *et al.*, 2002).

The extent of Mexico's terrestrial reserve network (6.9% of total land area) is about half of the global average (11.5%) and well below that of its neighbors Belize (36.6%), Guatemala (22.1%), and the United States (24.7%) (World Database on Protected Areas Consortium, 2003). Most of Mexico's reserve system is managed mainly for sustainable uses (8.7 million ha) compared with 1.2 million ha managed principally for conservation (World Database on Protected Areas Consortium, 2003). Although many of these areas maintain high levels of biodiversity when they are actively managed, external impacts can undermine local management, resulting in biodiversity losses.

The country's existing reserve network falls short of adequately protecting its extraordinary species diversity and endemism (Cantu, Wright, Scott, & Strand, 2004). Fully 12.7% of all species of mammals, amphibians, turtles, and birds that occur in Mexico are not protected—marking them as *gap species.* Moreover, the national reserve network does not cover 32.6% of the endemic species and 48.5% of the globally threatened species occurring in Mexico, with 55.5% of all globally threatened species endemic to Mexico (117 species) not covered in any part of their ranges. The extinction of any such species would diminish not only global biodiversity, but represents a significant loss to Mexico's historical and cultural patrimony as well.

3. DATA AND METHODS FOR LOCATING NEW PROTECTED AREAS

Expanding the current reserve system to maximize the number of representative species, habitats, and ecosystems in a manner that is both efficient and effective poses a tremendous challenge to conservation planners (Pressey, Humphries, Margules, Vanewright, & Williams, 1993). Unfortunately, many reserve systems throughout the world are highly biased toward particular subsets of natural features, usually habitats with less economic value and fewer species, while more biologically rich areas are inadequately protected (Pressey, 1994). Although individual reserves may be valuable, existing reserve networks often fail to represent the biodiversity within a particular region. Consequently, one can overestimate the biological value of a reserve system by focusing solely on its geographic extent (Pressey & Cowling, 2001). There is a general agreement that given the challenges of creating, financing, and adequately managing reserves, it is both socially and politically expedient, as well as financially necessary to minimize the overall size of reserve systems in any given country.

Approaches based on the *complementarity principle* (Vane Wright, Humphries, & Williams, 1991) explicitly assume that the aim is to produce a reserve network that, all together, can assure the preservation of a maximum of biodiversity elements or features (such as species, communities, land systems). The conservation value of any individual site is, therefore, the extent to which it complements the other sites in the network by contributing to the achievement of the conservation goals predefined for the network.

The recent availability of several georeferenced digital datasets makes it possible to identify potential reserve locations that meet complementarity requirements, thus meeting stated conservation goals. Other, socioeconomic data enable the evaluation of the human context of potential new reserves—simply stated, the number of people present, their geographic distribution, and socioeconomic and sociocultural characteristics—that provide key insights into the feasibility of potential new reserves. We briefly discuss the data and methods necessary for such an analysis in the following two sections.

(a) *Data*

Biological data for this study combined three global datasets showing the extent of occurrence (represented as mapped polygons) of breeding birds (799 species), mammals (427 species), and amphibians (346 species) within Mexico (IUCN-SSC & CI-CABS, 2003; Patterson *et al.*, 2003; Ridgely *et al.*, 2003). Each of these three data sources linked global databases to local expertise within Mexico to approximate the existing geographic occurrence of species. The resolution of the species data limits the resolution of the overall study to 1/4 deg grid cells, about 27 km to a side at the equator.

We obtained data on existing reserves in Mexico from official information submitted in 2003 to the World Database on Protected Areas Consortium by the Mexican Commission for Knowledge and Use of Biodiversity, the Mexican government agency responsible for monitoring and protecting biodiversity (World Database on Protected Areas Consortium, 2003). This dataset includes 223 reserves in Mexico. A 1/4 deg grid cell was considered *protected* if 20% or more of its area overlapped a protected area. There are 300 such cells.

Georeferenced estimates of agricultural suitability were generated by the recently concluded Global Agro-Ecological Zone (GAEZ) assessment (Fischer, van Velthuizen, Shah, & Nachtergaele, 2002). The GAEZ project developed a global *land resource database* combining soil, terrain, and climate data, to which it matched 154 crop, fodder, and pastureland *utilization types*. The latter, compiled for high (commercial), intermediate (mixed commercial and subsistence), and low (subsistence) input-management scenarios, enabled GAEZ researchers to identify productivity estimates for a broad range of agricultural strategy–crop combinations across the globe. By matching land utilization type requirements with land resources, the GAEZ assessment produced a global database of agronomically attainable yields by grid cells for a 5′ latitude/longitude raster (about 9 km to a side). For each input-management scenario, it estimated the suitability for individual crops and groups of crops in eight categories, ranging from very high to unsuitable, based on the percent of observed maximum yield for a particular crop category/input-management-scenario combination. [3] Our analysis used the GAEZ results for the maximum yield attainable with high inputs and irrigation and the maximum suitability of six different crop categories (cereals, cotton [as a surrogate for fiber crops], oil crops, pulses, roots and tubers, and sugar crops), providing the highest possible suitability for each 5′ grid cell in Mexico. This approach is conservative to give the best case scenario, almost certainly overstating productivity potential of a given cell were it actually used for crop production. It does not include the potential for certain perennials, including tree crops, often important for household economies.

Information on the human context drew upon three datasets: results of the 2000 Mexican census of population and housing (Instituto Nacional de Estadística, Geografía, e Informática, 2002); the global human footprint map of human presence, land use, and access (Sanderson *et al.*, 2002); and the most recent global land cover categorization (Latifovic, Zhu, Cihlar, & Giri, 2003). Population data for more than 197,000 individual communities in Mexico appear as points. Although these points do not show the precise spatial extent of any settlement, they are generally adequate for the vast majority of places containing relatively few people and thus covering limited geographic areas.

The human footprint data set uses a combination of measures of human presence and impact, including habitat conversion, access *via* roads and rivers, electrical power, and population to characterize *remoteness* (Sanderson *et al.*, 2002). The footprint analysis combined these variables into a continuous measure indicating relative human influence. Areas where this value was zero were considered to be outside the human footprint—that is, low human presence, limited access, and virtually no infrastructure nearby. They should not be interpreted to mean that there are no people, but that the evident impacts of human presence and land conversion are low or absent, and perhaps intentionally managed in ways that contribute to biodiversity conservation (e.g., Alcorn, 1989). Similarly, they do not indicate the abundance of wildlife or status of biodiversity conservation and could potentially be an *empty forest* (Redford, 1992). The 10% of each biome least affected by humans, in contrast, was deemed the "last of the wild." Finally, data on land use drew upon the Global Land Cover 2000 project. That project employed satellite imagery at a spatial resolution of 1 km from the 2000-growing season to categorize the entire earth's surface into 35 land cover classes. [4]

(b) *Methods*

We began this study by investigating how many of the vertebrate species analyzed were not covered (i.e., their ranges did not overlap with) by protected cells, as defined above, thereby identifying *gap species* (of which there are 261). [5] We then used a general integer linear programming model to find a complementary set of 1/4 deg cells representing each of these species at least once. Operations research techniques such as integer linear programming have been applied for several years to find optimal solutions to complementary reserve selection problems (Rodrigues, Cerdeira, & Gaston, 2000). The challenge of finding the minimum set of sites such that each of the 261 gap species is represented at least once is known in linear programming as the *set covering problem* (Ando, Camm, Polasky, & Solow, 1998; Balas & Ho, 1980; Padberg, 1979; Underhill, 1994). The objective function and associated constraints for this problem can be written as follows:

$$\text{minimize} \quad \sum_{j=1}^{n} x_j \tag{1}$$

$$\text{subject to} \quad \sum_{j=1}^{n} a_{ij} x_j \geqslant 1, \quad i = 1, 2, \ldots, m, \tag{2}$$

$$x_j \in \{0, 1\}, \quad j = 1, 2, \ldots, n, \tag{3}$$

where n is the number of sites, m is the number of gap species, a_{ij} is 1 if species i is present in site j and 0 otherwise, and x_j is 1 if and only if site j is selected.

Minimizing the objective function (1) subject to the stated constraints identifies the smallest number of grid cells that cover the 261 gap species. The minimum set to represent each gap species at least once consists of 94 cells. However, as with many problems of this type (e.g., Arthur, Hachey, Sahr, Huso, & Kiester, 1997), there may be a large number of sets of 94 cells that are equally optimal solutions for this problem. To address the potential conflict between reserves and rural livelihood needs, we selected among these possible solutions to identify the set of gap cells with the minimum overall agriculture suitability value across the 94 cells. [6] This was accomplished by solving the following integer linear programming problem:

$$\text{minimize} \quad \sum_{j=1}^{n} s_j x_j \tag{4}$$

$$\text{subject to} \quad \sum_{i=1}^{m} y_i \geqslant 261, \tag{5}$$

$$\sum_{j=1}^{n} x_j \leqslant 94, \tag{6}$$

$$\sum_{j=1_i}^{n} a_{ij} x_j \geqslant y_i, \quad i = 1, 2, \ldots, m, \tag{7}$$

$$x_j \in \{0, 1\}, \quad j = 1, 2, \ldots, n, \tag{8}$$

$$y_i \in \{0, 1\}, \quad i = 1, 2, \ldots, m, \tag{9}$$

where s_j is the agricultural suitability of site j, y_i is 1 if gap species i is covered and 0 otherwise, and the other symbols are as above.

Minimizing the objective function (4) subject to the stated constraints yields the smallest collection of grid cells that cover the 261 gap species while minimizing agricultural suitability. This forces the solution toward gap cells away from areas desirable for their agricultural potential.

Once we defined this set of 94 gap cells, we overlaid other datasets to evaluate the human population and land-use context within the selected cells. Estimating population involved identifying those communities occurring in a cell and summing their 2000 population. Examining land use for the selected cells, in turn, involved calculating the amount of various land cover categories occurring in selected cells and comparing the extent of certain types of land cover (those involving agriculture or urbanized areas) to the total area of each selected cell.

4. RESULTS

Figure 1 presents the set of 94 gap cells needed to cover all species of amphibians, breeding birds, and mammals not represented within Mexico's existing reserve system. These cells are the geographic representation of the minimum complementary set representing all gap species in nonprotected cells with limited

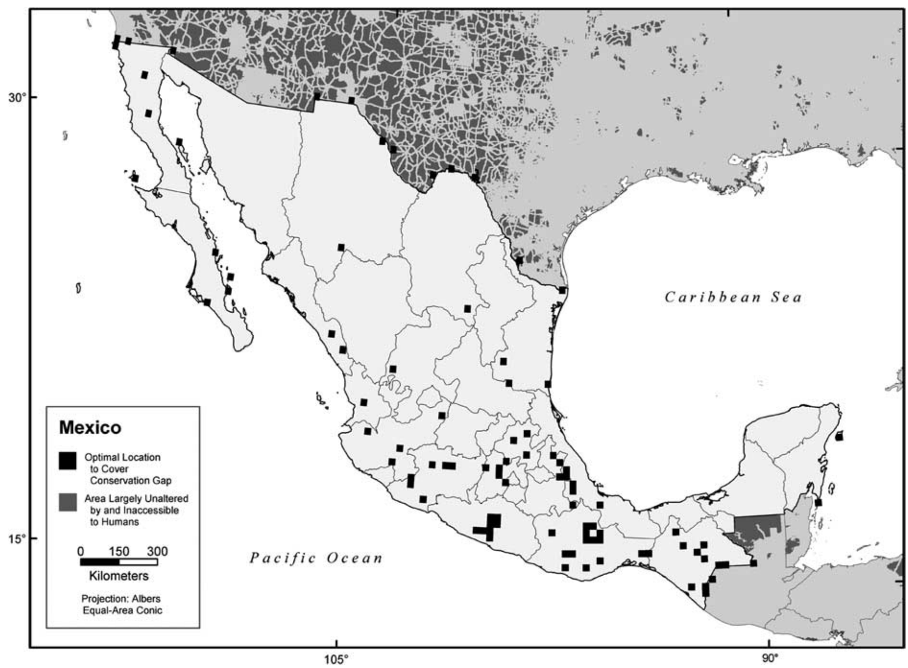

Figure 1. *Map of gap cell locations in Mexico and the human footprint.*

agricultural suitability. As explained above, the 94 gap cells shown represent one possible solution; other solutions selecting a different set of at least, but not fewer, than 94 cells are also possible.

There are several caveats to the results of our analysis. First, the cells selected over-represent the area needed because species at the edges of their range between Mexico and the United States, Mexico and Guatemala, and Mexico and Belize may appear as rare and unprotected in Mexico. Although this indeed might be the case in Mexico, the same species might be more common in neighboring countries. This bias can be corrected through considering the additional ranges of selected species, but we did not do this, as the aim of this study was to identify country-specific solutions that do not rely on decision making in other nations. Second, the gap selection methodology does not consider information on the habitat requirements needed for each species (e.g., whether species are migratory species or wide-ranging species, the type of habitat needed, or how species are affected by disturbance) or on the adequacy of areas to conserve them (Cowling & Pressey, 2003; Pressey, 1998). This information is essential before one can determine that the set of gap cells selected for inclusion into the reserve system is ecologically viable.

We also distinguish between all gap cells and the 71 gap cells with forests as the primary ecosystem type. For 23 gap cells, the primary ecosystem type is a desert or other arid areas dominated by shrubs. These cells included many located along the border with the United States (including several affected by the selection bias noted above), along with the gap cells in Baja California. From a general conservation standpoint, it is important to stress that representation for all species would certainly include areas that consisted of shrubland and desert, in addition to the 71 forested gap cells.

Given that this set of gap cells represents one possible way of completing the Mexican reserve network to cover the country's diversity of mammals, amphibians, and breeding birds, how feasible is management for biodiversity conservation within them?

(a) *Reserve selection and agricultural productivity*

We selected agricultural productivity as a proxy for places with potential to support rural livelihoods. A paramount concern given the increasing numbers of rural poor and the growing demands for additional land for food production is to minimize the amount of good agricultural land in reserves (McNeely & Scherr, 2002; Wood, Sebastian, & Scherr, 2000). Questions have arisen as to whether the

> expanding area of tropical forest reserves represent the alienation of large areas of potential agricultural land ... or reduce the ability of developing countries either to compete in agricultural export markets ... or to reach and maintain self-sufficiency in staple foods ... (Wood, 1995, p. 121).

In the case of Mexico, existing reserves have soil types ranging from medium to poor in productivity (Cantu *et al.*, 2004), a conclusion supported by the present study (Figure 2). Cells with a higher percentage of land under protection also had a higher percentage of land identified as unsuitable for agriculture: For example, 63% of the cells for all of Mexico with more than 60% of their area under protection emerged as unsuitable for agriculture. Note that this is considerably higher than for Mexico as a whole, where although highly productive land is not widespread, it is even rarer in locations containing reserves. These findings for Mexico's existing reserves echo global findings showing that reserve systems generally occur in areas with low agricultural potential (Brandon & Gorenflo, submitted for publication; Gorenflo & Brandon, 2005; Scott *et al.*, 2001). Such a conclusion makes intuitive sense, since in places with long-term human presence people tend to settle near more productive lands, leaving intact habitats in the places that people viewed as marginal.

Our analysis was intended to select among possible solutions to identify the set of gap cells with the minimum overall agriculture suitability to minimize the presence of good agricultural land. Figure 2 compares the 94 gap cells to all 3,040 cells that comprise Mexico as a whole, to the 556 cells that occur in the existing reserve system, to cells with variable percentages of the existing reserves, and to the forested gap cells. The comparison reveals that the majority of cells (52%) within Mexico are unsuitable to very marginal for agriculture—characterized by poor soil, undesirable topography (e.g., steep slopes), inadequate climatic conditions (e.g., inadequate rainfall), or some combination of these characteristics. Such results certainly do not mean that people will

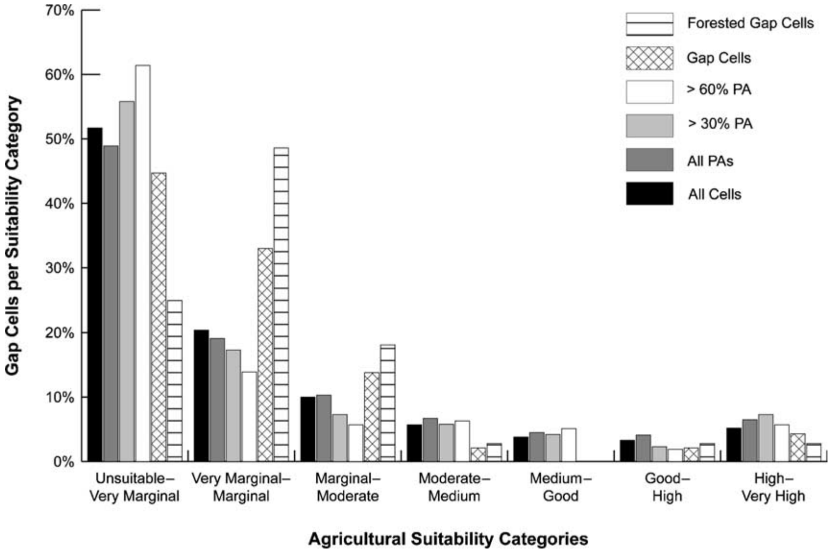

Figure 2. *Comparison of agricultural suitability for different land classifications.*

not use these areas for agriculture; indeed, many agricultural decisions by rural populations are based on land availability. But, it does mean that people depending on these lands for their livelihoods are likely to remain poor in the absence of other economic alternatives to support them beyond subsistence.

The three lowest categories of agricultural suitability range from unsuitable to marginal—the three lowest categories signify 25% or less of maximum potential productivity (see note 3). For these three lowest categories, more land within the gaps fell into these categories than for Mexico as a whole. The percentage of cells in the two least productive agricultural categories was 72% for all cells in Mexico, 78% for all gap cells, and 74% of all forested cells. Looking at the lowest four agricultural suitability categories that can be characterized as unsuitable to moderate, subsistence agriculture shows that 92% of the selected gap cells and 92% of the forested gap cells have these low levels of productivity. This can be compared to 82% for all of Mexico and 78% for existing reserves. Opportunities for commercial, intensive agriculture with high and highest yields are consistently lower in the gap cells and forested cells.

There are three areas in Mexico where large numbers of gap cells occur (see Figure 1). The first cluster (A) consists of seven contiguous cells in the state of Guerrero; the second group (B), in the state of Oaxaca, consists of eight cells midway between the Caribbean and the Pacific; and the third (C) cluster of eight cells consists of a linear arrangement straddling the states of Veracruz and Tlaxcala. Most of these 23 areas have low agricultural suitability, with a few notable exceptions. In cluster A, one cell reaches the marginal range; in cluster B, the cells show very low productivity—with the average value remaining in the unsuitable range. Some of the most productive cells occur in cluster C, with two of the cells in that group reaching a good range and the average for the eight cells being better than marginal productivity. The proximity of these cells to one another suggests that apart from creating reserves within them, the potential exists to manage them as conservation corridors that improve conservation and support local livelihoods through zoning and the introduction of biodiversity-compatible land uses (Sanderson *et al.*, 2003).

The agricultural suitability of gap cells and forested gap cells tends to be low, both in absolute terms and when compared to elsewhere in Mexico, and within the existing reserve system. The low level of productivity of all gap cells, especially forested gap cells, suggests that reserve expansion within gap cells that can support sustainable management of existing resources is likely to maintain ecosystem functions, support biodiversity, and over the long term, provide more lasting benefits to rural residents.

(b) *Reserve selection and human influence and population*

Within the Mexican context, our analysis suggests that the options to locate reserves far away from people while meeting conservation goals are virtually nonexistent. The human footprint in Mexico is geographically extensive. Only six of the 94 gap cells lie partially within places characterized as the *wildest of the wild*; no gap cells are completely within the wildest areas. Fifteen gap cells lie at least partially outside the human footprint, and only one is entirely outside the footprint. The remaining 78 gap cells are located in places where roads or navigable rivers, power lines, and people occur.

Increased concentration of people often also concentrates human impacts geographically, although the absence of dense populations does not guarantee an absence of impacts. Biodiversity can be well managed, or heavily impacted, by the actions of relatively few people (Gorenflo, 2002, in press). Social assessments and participatory planning are needed to define what existing and potential roles resident or immigrant populations can have in managing large multiple use areas (Brandon *et al.*, 1998). However, other data can provide insights into the likely complexity of designing reserves that limit the number of people included within. Analysis of human presence with more detailed population data reveals that more than half of the 94 cells contain more than 10,000 people (Figure 3). Although this seems like a substantial number of people, a total of 10,000 (for instance) translates into only about 13.7 persons per km^2. The demographic data used for this analysis provide information on community size as well as location within each cell, enabling one to examine the degree to which the human population was concentrated geographically or dispersed in sufficient numbers to increase the challenge to conservation. For example, the most northwestern gap cell contains Tijuana, a city of more than a million people located near the western edge of the cell. Subtracting Tijuana from the cell leaves a total population of fewer than 1,000, demonstrating that it might be easier to work in the area outside of Tijuana than one would expect based on data for the whole gap cell.

When the population residing in communities of more than 5,000 people is subtracted from the total cell population, 55 have fewer than 10,000 people, and 19 of those have fewer than 1,000 people. Such analysis shows that in most gap cells, new reserves could be designed to exclude more populated communities. In the case of forested gap cells, 54 contained pop-

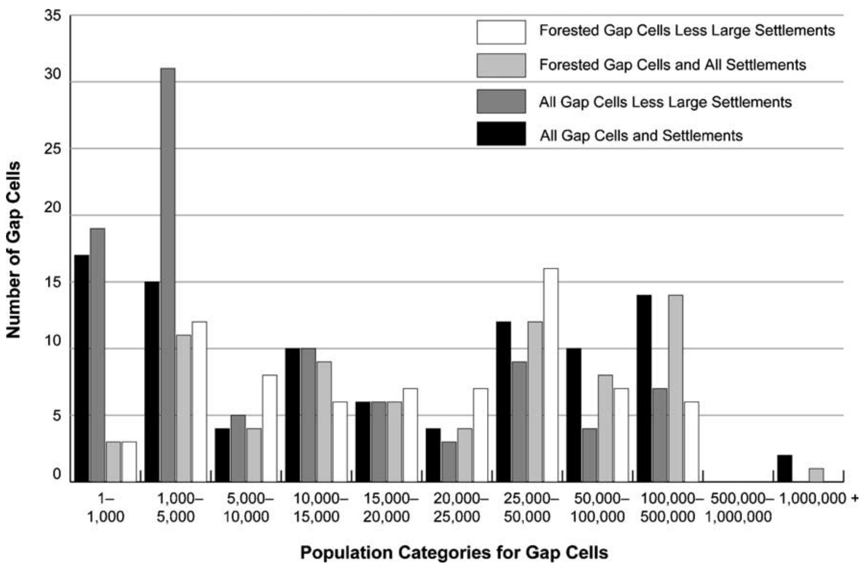

Figure 3. *Human population in gap cells.*

ulations greater than 10,000, only declining slightly to 49 when towns greater than 5,000 were removed. This indicates that virtually all gaps have dispersed human presence although it may be sparse in some places. Generally, sparse populations provide greater opportunities for coordinated conservation planning and development planning, and creates the possibility for mechanisms that directly benefit rural residents through livelihood support for traditional reserve management activities, including boundary demarcation, zoning, tourism, ecological monitoring, and restoration activities.

(c) *Reserve selection and land uses*

Land cover data were used to calculate the percentage of each gap cell with land cover that is potentially compatible with biodiversity conservation—that is, habitat that is not urban, agricultural, or otherwise converted from a natural state. Although several gap cells contained populations in excess of 25,000, habitat conversion appears to be spatially restricted: Only 20 of the 94 gap cells featured land cover incompatible with conservation over more than half their land area (Figure 4). Of that subset, eight had more than 80% of their respective land area transformed from natural habitat to other uses.

Forests covered 46 of the gap cells over at least half of their land area. Of these 46 predominantly forested cells, an average of only 19% of their land area was incompatible with conservation, although incompatibility coverage ranged from 0% to 46% across all forested gap cells. The cells that were at least half-forested also tended to have large human populations: Eleven cities with more than 100,000 inhabitants occurred in more heavily forested gap cells, as opposed to five cities with similarly large populations in areas with a few forests.

Although the presence of people indicates potential adverse impacts, the coexistence of cities in gap cells with considerable remaining forest area indicates a strong potential for human residence near reserves, as long as the capacity exists to manage reserves and support biodiversity-friendly economic activities within and around reserves.

One important element to consider in conservation planning is the degree to which compatible land uses are adjacent to one another rather than fragmented (Sanderson *et al.*, 2003). The amount of area needed by different

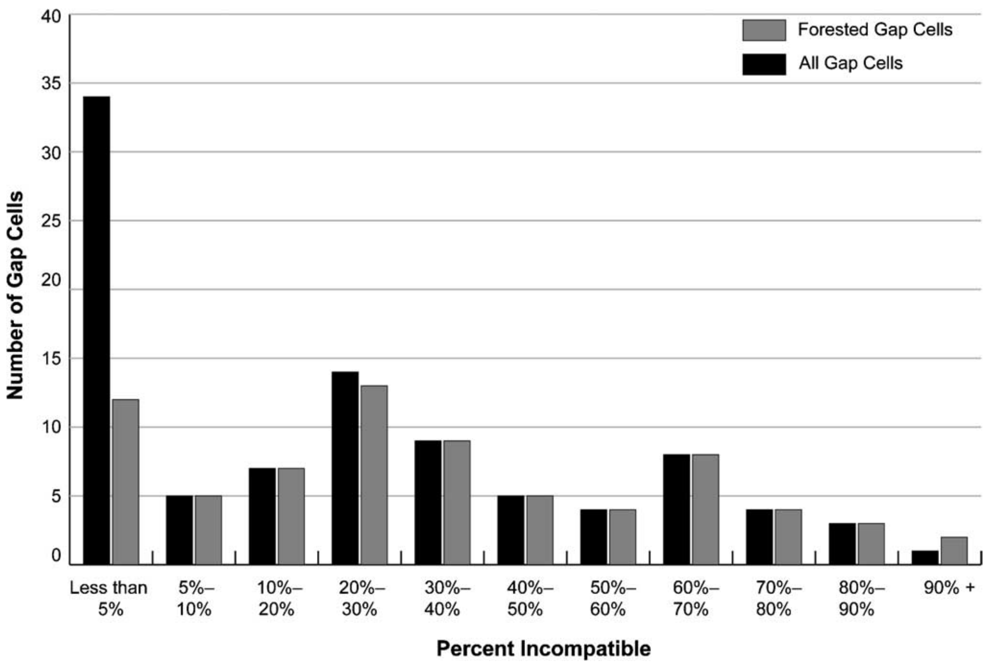

Figure 4. *Land use incompatible with conservation.*

species is highly variable. We arbitrarily took 10,000 ha as indicative of contiguous areas that might be of a sufficient size to act as core areas and maintain viable wildlife populations. Of the 94 cells identified in our gap analysis, over half (51) had at least 10,000 contiguous hectares of unaltered habitat. Of the 72 forested gap cells, 21 have less than 10,000 contiguous hectares compatible with conservation, and an average of 64% of their land area is incompatible with conservation. The 51 cells containing at least 10,000 ha of contiguous forests appear to provide a good context for conservation, with an average of only 21% in uses that are incompatible with conservation.

Conservation will be challenging in about 10 forested cells and extremely difficult if not impossible in two of them. These latter two cells contain less than 2,000 ha in contiguous biodiversity-compatible land, with more than 88% of their respective land areas in uses that are incompatible with conservation efforts. In these two forested gap cells, conservation almost certainly would have to focus on restoration and support for compatible uses within these highly fragmented habitats.

All the three clusters of cells identified above (see Figure 1) have considerable contiguous unconverted habitat, with much of this land continuing between adjoining cells. In the Guerrero cluster (A in Figure 1), six of the seven cells contain more than 18,000 ha of habitat compatible with conservation, five of these with more than 33,000 ha of such habitat. In the Oaxaca cluster (B in Figure 1), seven of the eight cells contain more than 10,000 ha of contiguous unconverted habitat, with five of these featuring more than 30,000 ha of such land cover. In contrast, in the Veracruz–Tlaxcala cluster (C in Figure 1), only four of the eight cells had more than 10,000 ha of land cover compatible with conservation.

Half or more of the land area in 74 of the 94 gap cells comprised uses that are potentially compatible with conservation; half or more of the land area in 52 of the forested gap cells was compatible with biodiversity conservation. In all cases, protecting the remaining biodiversity will be most challenging in gap cells with less than 10,000 ha of contiguous natural habitat and where most land uses are incompatible with conservation. Habitat fragmentation in these cells already is substantial. Because species have very different reactions to human disturbance and different demands for uninterrupted habitat, whether these gap cells are viable for conservation will need closer examination. Although many of the gap species have survived with humans in relative proximity, a relationship exists between measures of human activity and species extinction risk (e.g., Brooks, Pimm, & Collar, 1997; McKinney, 2001). Therefore, when possible, creating large core reserves with few or no people within is the most desirable.

The ideal size, design, and management of a reserve is based on numerous factors, among them habitat requirements of the target species, adequacy of the area, and the socioeconomic context in and around proposed reserves (Brandon, 2002; Cowling & Pressey, 2003; Pressey, 1998). In ecological terms, larger protected areas are more desirable for long-term species conservation (Bierregaard *et al.*, 2001) and to maintain ecological and evolutionary processes (Cowling, Pressey, Lombard, Desmet, & Ellis, 1999). However, small reserves are adequate for some species and are almost always better than no reserve or management over an area (Turner & Corlett, 1996). Finer resolution spatial data and ground truthing are needed, along with extensive consultations, before anyone even begins to outline potential reserves on maps.

In general, reserves with few people around can be designated as "core" protected areas, while areas with residents throughout need to be established as multiple use areas. Reserve categories such as biosphere reserves or other sites with residents within can only be successful if there is participation and management of zoning and use designations, and enforcement that is locally generated and supported (Brandon, 2002; UNESCO, 2002).

Community-based conservation approaches may be reinforced to support management for gap species in places with traditional and indigenous peoples, or in the Mexican case, *ejidos* (Bray *et al.*, 2003). As part of these site consultations, it is vital to understand the social context to "establish the right kind of park in the right kind of place" (Brandon, 2002). This requires a solid understanding of the social context for new reserve establishment, and the right mechanisms for participation. Failure to do this will likely lead to losses of biodiversity and will undermine support for conservation at local, national, and international levels (Brandon *et al.*, 1998; Brechin, Wilshusen, Fortwangler, & West, 2002).

5. DISCUSSION

The analysis conducted in this study demonstrates a potential methodology to assure species conservation while minimizing the total area included in reserves and their agricultural potential. Ultimately, long-term conservation will succeed only when species are protected in reserve networks (and elsewhere, ideally) that can meet species needs while minimizing the opportunity costs of conservation for rural residents. Our findings demonstrate that location–allocation modeling is an efficient way to minimize the total area needed to cover currently unprotected species. CONABIO, the Mexican government agency responsible for monitoring biodiversity in that country, has proposed 151 priority areas for the creation of new reserves (Cantu *et al.*, 2004). There was some overlap between the CONABIO sites and the gap cells selected in this exercise; of all gap cells, 66% occur partially or totally in the CONABIO priority areas. Yet the total land area that is compatible with conservation in all 94 gap cells is only 3.7 million ha, about 7.1% of the 51.4 million ha encompassed by the CONABIO sites (Cantu *et al.*, 2004). In completing the global reserve system, it is vital to insure that all gap species are represented. But in countries such as Mexico, with large rural populations, and relatively high pressure on rural sector resources, it often is desirable for reserve systems to meet species needs while occupying as little rural land as possible for biodiversity conservation, although there are a great many other reasons to establish reserves (e.g., watershed and soil protection, scenic beauty, etc.).

Protected areas play a vital role in biodiversity conservation, yet they are not islands—they are components of their surrounding social and ecological contexts (Brandon *et al.*, 1998). There has been a lack of realism over what level of resource extraction is sustainable in sustainable use zones within reserves, and how these resources can be managed sustainably (Redford & Richter, 1999). It is probably unrealistic to expect that sustainable use is possible while also expecting the amounts harvested to be sufficient to lift rural residents from poverty (e.g., see DFID, 2002). Unfortunately, much of the debate about the opportunity costs imposed on rural residents by reserves ignores both the productive potential of reserve areas, and detracts from looking at the broader context and drivers that are at the core of rural poverty. Increasing the footprint of reserves is ideally done through corridor and regional-scale planning that integrates reserves into a matrix of surrounding land uses that are biodiversity friendly. While there are renewed efforts to strengthen protected area management (including community-conserved areas), conservationists are increasingly looking at land uses beyond reserve boundaries, at corridor, ecoregional, and landscape scales to identify how strategies that support conservation and rural livelihoods might be better addressed away from protected areas.

This paper demonstrates that the productive agricultural potential in many proposed reserve areas is low. Yet the results of our analysis do not imply that poor agricultural suitability for potential new reserves will lead to reduced subsistence agricultural pressure; this will occur only if there is a reorientation of development policies affecting adjacent lands that do not lead to crop production on land poorly suited for that purpose. Failure to focus these development investments in ways that also support conservation leads to reduced utility of marginal lands for conservation, destruction of key (and often valuable) ecosystems services, and (for many rural residents) lives of poverty on fragile lands with low productivity (Angelsen & Kaimowitz, 2001; Brandon, 2000).

Supporting livelihood alternatives that are compatible with conservation will depend in part on the species requirements of a given area. For example, reserves created to conserve birds may benefit from shade coffee systems while the same systems may provide no benefit for amphibians needing wetlands. When conditions permit, agriculture that is compatible with conservation should be emphasized on land in and around reserves, especially when there is no market access. *Ecoagriculture* is the general term used for land-use systems, both traditional and of recent origin, that simultaneously support both conservation and food production (McNeely & Scherr, 2002). Other compatible development solutions may involve activities associated with conservation itself, such as park management, monitoring, and ecotourism.

Unfortunately, the options often are limited to the extraction of some sort of resource in the proximity of reserves, promoting an activity often inconsistent with conservation within and outside the boundaries of reserves. In such cases, the challenge is to support rural communities in expanding their roles as resource stewards and managers in addition to any support for development.

The threats to communal lands in rural Mexico are similar to the challenges faced by communal lands and reserves globally: They appear to be open access due to "the conflicts of boundary demarcation; internal divisions of mixed-indigenous and nonindigenous communities associated with conflicts around social organization, and cultural and ethnic issues; and conflict of interests over management" (Segura, 2000, p. 10). Encouraging resource stewardship might include involvement of local communities in participatory boundary demarcation and zoning for reserves. This helps residents establish core reserves for species protection, maintain large intact habitats for ecosystem services, and provides residents with a framework for enforcement to exercise control over both large and small scale threats to their lands and resources (Brandon, 2002; Sanderson, 1998).

The protection of biodiversity need not always occur through nationally designated reserve systems. In the Mexican context, indigenous or rural communities have had a role in maintaining land uses that are compatible with conservation, while other parts of Mexico have experienced the loss of natural habitat. Identifying those communities with land and resource rights in the contiguous forested gap cells offers a relatively direct mechanism to engage people in conservation. One priority in the Mexican context would be to support conservation in the 51 gap cells with 10,000–50,000 ha of contiguous natural habitats. Supporting these groups in efforts to manage lands for global biodiversity needs and ecosystem services is likely to have a much greater return to them than transforming additional land to agricultural uses (Bray *et al.*, 2003). Support and strengthening of existing management institutions and land tenure systems may be necessary; assumptions that existing community organizations can appropriately manage large areas may be naïve (Agrawal & Gibson, 1999; Barrett, Brandon, Gibson, & Gjertsen, 2001; Gibson, McKean, & Ostrom, 2000). However, land availability for agriculture amid population growth remains a concern. One study in Mexico found a positive association between the population size of *ejidos* and environmental degradation, arguing that this was indicative of the complexity of managing rural sector reforms and the resource base simultaneously (Winters, Davis, & Corral, 2002).

Recent analyses for Mexico indicate that strategic investments can make a difference, and can lift people out of poverty even when they only have small amounts of land. To achieve these large income gains for the rural poor, it is necessary to have both the complementary assets (such as education) and the contextual settings (such as infrastructure) in place (Finan, Sadoulet, & de Janvry, 2002). These findings were particularly significant for nonindigenous small farmers with a primary school education and road access: Additional land provided seven times the welfare benefit than for farmers without these assets.

The feasibility of agricultural solutions of course depends on the suitability of the place. Although poor agricultural suitability characterized most gap cells and forested gap cells identified in our analysis, a few occurred in locations with *high* or *very high* suitability. Intensifying agriculture on better lands can make major contributions to poverty reduction and reduce the conversion of lands unsuitable for agriculture (Lee & Barrett, 2000). Intensification on highly productive lands should be encouraged, although the type and level of inputs used to increase productivity should be carefully monitored to bring about a *doubly green revolution* (Conway, 1997).

Without abandoning those people who reside near reserves, efforts to improve rural lives and livelihoods may be better supported in places that are less marginal. In Mexico as elsewhere, the location of poor agriculturalists on marginal lands fails to improve livelihoods, and leads to increased deforestation—diminishing possibilities for future residents of these areas (Deininger & Minten, 2002). Investments that provide small amounts of land, but with access to markets, education, and healthcare for small landholders is likely to be better over the long term for promoting stable lives and livelihoods. Paved roads providing market access for agricultural production are also positively correlated with deforestation (Chomitz & Gray, 1996; Deininger & Minten, 2002). Yet such investments require thoughtful planning, since increased revenue from intensified agriculture has led to expansion of agriculture at the cost of forests (Mellor, 2002). For these reasons, investments in roads and agricultural improvements should be directed away from areas with low agricultural productivity but high-biodiversity value. This relates to national development scenarios. It is worth determining how land with the highest productive potential is used in Mexico, who owns and manages it, and if it is used to its highest potential.

6. CONCLUSIONS

Mexico is a country of global importance for biodiversity, yet there are many species not protected by Mexico's existing reserve system. Mexico is also a country that needs to address rural poverty and promote rural development, aims that potentially could conflict with any attempts to set aside additional land for conservation. This study has presented a method that can identify a minimum number of reserves to cover conservation gaps in Mexico, located on lands with the lowest agricultural productivity. Reducing the amount of land with high agricultural potential in reserve systems can help decrease conflict between conservation and rural livelihoods. In addition to reducing the overall socioeconomic and opportunity costs of establishing and maintaining reserve networks, such methodologies represent an explicit attempt to incorporate development concerns. Selecting sites with low agricultural productivity guarantees that conservation uses land undesirable for crop production. Because agriculture will never provide secure livelihoods in these places, investments in alternative sources of income, (e.g., direct support for management of high-biodiversity areas, tourism, agroforestry) is likely to provide more secure futures.

The 94 gap cells identified in this paper represent one possible solution to the inadequacies in Mexico's current system of reserves, intended to identify the advantages of simultaneously considering conservation and development. It should not be presumed that we are making final recommendations that these are the exact sites that should be added to Mexico's reserve network. The analysis does demonstrate that even in places such as Mexico where conservation may be viewed as very challenging to implement, opportunities still abound. In most of the 94 gap cells selected, and particularly within the forested gap cells, there were substantial amounts of land in uses compatible with conservation, human residence was low, and agricultural suitability was low compared to national levels. Unfortunately, actual decisions on which land to use for agriculture often do not rest on growing a particular crop in the most suitable place for it. For this reason, investments and policy reforms are needed to support rural livelihoods on land with the capacity to meet rural agricultural demands, rather than allowing crop production on land that perpetuates poverty while compromising biodiversity. In places with very low human presence, it may be possible to create core protected areas, where biodiversity conservation is the primary objective. In populated places where land uses are incompatible with conservation, zoning at finer scales and regional or corridor scale planning will be needed.

Increasingly, conservation planning and implementation must occur as part of comprehensive planning efforts that consider local, regional, and national development strategies. Our results demonstrate that reserve selection that first incorporates biological criteria, and then supplements it with data on the rural sector, in this case agricultural suitability, can help planners meet conservation goals without substantial conflict with current human settlement, land use, or future agricultural development. Such studies can highlight places with high agricultural potential, where investments located away from high-biodiversity areas can support livelihoods that are more secure and with higher economic returns.

Perhaps the most significant challenge facing both conservation and development is the need to support rural livelihoods by adequately assessing and capturing the value of environmental services (Kremen *et al.*, 2000). The rural poor may have few alternatives when soils erode, water quality and quantity are diminished, and forest resources disappear. It is therefore in their long-term interest to manage these resources well. However, markets rarely recognize or reward resource owners for the host of services generated by natural ecosystems that are beneficial to society—e.g., carbon storage, watershed protection, pollination, nutrient cycling, and decomposition—resulting from both consumptive and nonconsumptive (e.g., ecotourism) uses of biodiversity. Furthermore, many of these benefits are realized at a global scale (Kremen *et al.*, 2000). If communities obtained even part of the value of protecting forest resources, there would be a double benefit: Rural livelihoods could be directly supported for restoring and managing biologically important areas. Ultimately, improving rural livelihoods in places with high biodiversity will require both better planning and compensation, from the international community in recognition of the global benefits provided by local-level resource managers. For these reasons, it is appropriate that the costs of implementing or maintaining conservation at local scales be shifted to national and global scales, and be financed by multi and bilateral institutions, foundations, and private corporations and individuals (Balmford & Whitten, 2003).

NOTES

1. The term "protected area" is often erroneously equated with national parks that impose limits on all extractive and consumptive human uses, yet the World Conservation Union (IUCN) classification system only limits human activities in two of the six reserve categories (IUCN, 1994). Globally, only a small amount of land area is managed strictly for biodiversity conservation.

2. Species limited to a restricted geographic area.

3. Agricultural suitability categories are as follows: *Very high* (>85% of maximum yield), *high* (70–85% of maximum yield), *good* (55–70% of maximum yield), *medium* (40–55% of maximum yield), *moderate* (25–40% of maximum yield), *marginal* (5–25% of maximum yield), *very marginal* (suitability index = 0–5% of maximum yield), and *unsuitable* (suitability index = 0% of maximum yield).

4. Land cover data cannot distinguish between natural grasslands and pasture derived from satellite imagery; potentially overestimating compatible land cover for areas with grasslands and grazing.

5. Results of a global gap analysis were published recently, describing an analytical effort designed to identify conservation gaps for the entire world (Rodrigues *et al.*, 2004). We conducted the present study focusing exclusively on Mexico in part to expand the gap analysis methodology to consider human context, and in part to focus on a single national entity—the level where decisions about conservation ultimately are made.

6. The resolution of the analysis to identify gap cells was 1/4 deg, the resolution of the species occurrence data available. The resolution of the agricultural suitability estimates was 5′ (1/12 deg), which we aggregated to 1/4 deg to enable the identification of gap cells with minimal agricultural suitability.

REFERENCES

Agrawal, A., & Gibson, C. K. (1999). Enchantment and disenchantment: the role of community in natural resource conservation. *World Development, 27*(4), 629–649.

Alcorn, J. (1989). An economic analysis of Huastec Mayan agroforestry management. In J. G. Browder (Ed.), *Fragile lands of Latin America: Strategies for sustainable development* (pp. 182–206). Boulder, CO: Westview Press.

Ando, A., Camm, J., Polasky, S., & Solow, A. (1998). Species distributions, land values and efficient conservation. *Science, 279*, 2126–2128.

Angelsen, A., & Kaimowitz, D. (Eds.) (2001). *Agricultural technologies and tropical deforestation*. Wallingford, UK: CAB International.

Arthur, J. L., Hachey, M., Sahr, K., Huso, M., & Kiester, A. R. (1997). Finding all optimal solutions to the reserve site selection problem: formulation and computational analysis. *Environmental and Ecological Statistics, 4*, 153–165.

Balas, E., & Ho, A. (1980). Cutting planes from conditional bounds: a new approach to set covering. *Mathematical Programming Study, 12*, 19–36.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R. E., *et al.* (2002). Economic reasons for conserving wild nature. *Science, 297*, 950–953.

Balmford, A., & W__tten, T. (2003). Who should pay for tropical conservation, and how could the costs be met? *Oryx, 37*, 238–250.

Balvanera, P., Daily, G. C., Ehrlich, P. R., Ricketts, T. H., Bailey, S. A., Kark, S., *et al.* (2001). Conserving biodiversity and ecosystem services. *Science, 291*, 2047.

Barrett, C. B., Brandon, K., Gibson, C., & Gjertsen, H. (2001). Conserving tropical biodiversity amid weak institutions. *BioScience, 51*(3), 497–502.

Bierregaard, R. O., Jr., Laurance, W. F., Gascon, C., Benitez-Malvido, J., Fearnside, P. M., Fonseca, C. R., *et al.* (2001). Principles of forest fragmentation and conservation in the Amazon. In R. O. Bierregaard, Jr., C. Gascon, T. E. Lovejoy, & R. C. G. Mesquita (Eds.), *Lessons from Amazonia: The ecology and conservation of a fragmented forest* (pp. 371–386). Yale University Press.

Brandon, K. (2000). Moving beyond Integrated Conservation and Development Projects (ICDPs) to achieve biodiversity conservation. In D. R. Lee & C. Barrett (Eds.), *Tradeoffs or synergies? Agricultural intensification, economic development and the environment*. Wallingford, UK: CAB International.

Brandon, K. (2002). Getting the basics right: key actions in designing effective parks. In J. Terborgh, L. C. Davenport, & C. Van Schaik (Eds.), *Making parks work: Identifying key factors to implementing parks in the tropics* (pp. 443–466). Covelo, CA, Washington, DC: Island Press.

Brandon, K., & Gorenflo, L. J. (submitted for publication). *Agricultural suitability in tropical hotspots*.

Brandon, K., Redford, K., & Sanderson, S. (Eds.) (1998). *Parks in Peril: People, politics, and protected areas* (pp. 415–440). Covelo, CA, Washington, DC: Island Press.

Bray, D., Merino, L., Negreros, P., Segura, G., Torres, J. M., & Vester, H. (2003). Mexico's community managed forests as a global model for sustainable landscapes. *Conservation Biology, 17*(3), 672–677.

Brechin, S. R., Wilshusen, P., Fortwangler, C., & West, P. C. (2002). Beyond the square wheel: toward a more comprehensive understanding of biological diversity conservation as social and political process. *Society & Natural Resources: An International Journal, 15*, 41–64.

Brooks, T., Pimm, S. L., & Collar, N. J. (1997). Deforestation predicts the number of threatened birds in Insular Southeast Asia. *Conservation Biology, 11*, 382–394.

Bruner, A. G., Gullison, R. E., Rice, R. E., & Fonseca, G. A. B. (2001). Effectiveness of parks in protecting tropical biodiversity. *Science, 291*, 125–128.

Cantu, C., Wright, R. G., Scott, J. M., & Strand, E. (2004). Assessment of current and proposed nature reserves of Mexico based on their capacity to protect geophysical features and biodiversity. *Biological Conservation, 115*, 411–417.

Chomitz, K. M., & Gray, D. A. (1996). Roads, land use, and deforestation: a spatial model applied to Belize. *World Bank Economic Review, 10*(3), 487–512.

Conway, G. (1997). *The doubly green revolution*. Ithaca, NY: Cornell University Press.

Cowling, R. M., & Pressey, R. L. (2003). Introduction to systematic conservation planning in the Cape Floristic Region. *Biological Conservation, 112*, 1–13.

Cowling, R. M., Pressey, R. L., Lombard, A. T., Desmet, P. G., & Ellis, A. G. (1999). From representation to persistence: requirements for a sustainable system of conservation areas in the species-rich Mediterranean-climate desert of southern Africa. *Diversity Distributions, 5*, 51–71.

Deininger, K., & Minten, B. (2002). Determinants of deforestation and the economics of protection: an application to Mexico. *American Journal of Agricultural Economics, 84*(4), 943–960.

Department for International Development (DFID) (2002). Wildlife and poverty study. Livestock and Wildlife Advisory Group, Department for International Development, London.

Finan, F., Sadoulet, E., & de Janvry, A. (2002). *Measuring the poverty reduction potential of land in rural Mexico*. University of California at Berkeley, Available fromhttp://are.berkeley.edu/~sadoulet/papers/Landpaper_WP.pdf.

Fischer, G., van Velthuizen, H., Shah, M., & Nachtergaele, F. (2002). *Global agro-ecological assessment for agriculture in the 21st century: methodology and results*. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Gibson, C., McKean, M., & Ostrom, E. (Eds.) (2000). *People and forests: communities, institutions, and governance*. Cambridge, MA: MIT Press.

Gorenflo, L. J. (2002). The evaluation of human population in conservation planning: an example from the Sonoran Desert Ecoregion. *Publications for Capacity Building*, The Nature Conservancy. Arlington, VA.

Gorenflo, L. J. (in press). Human demography, land use, and conservation in the Apache highlands ecoregion, US–Mexico borderlands. In R. Cincotta, & D. Macgeean (Eds.), *Demographic ecogeography: Relationships between human population dynamics and biological diversity*. Berlin: Springer-Verlag.

Gorenflo, L. J., & Brandon, K. (2005). Agricultural capacity in high biodiversity forest ecosystems. *Ambio, 34*, 199–204.

Hilton-Taylor, C. (2000). *IUCN red list of threatened species*. Gland, Switzerland, Cambridge, UK: IUCN/SSC.

Human Development Index, Mexico. United Nations Development Program. Available from http://www.undp.org/hdr2003/indicator/cty_f_MEX.html.

Instituto Nacional de Estadística, Geografía, e Informática (2002). Localidades de la Republica Mexicana. Aguascalientes: Instituto National de Estadística, Geografia, e Informática. Editado por Comision Nacional para el Conocimiento y Uso de la Biodiversidad.

IUCN (1994). *Guidelines for protected area management categories*. Gland, Switzerland: IUCN.

IUCN-SSC, & CI-CABS (2003). *Global amphibian assessment*. Gland: IUCN; & Washington DC: Conservation International.

Klooster, D., & Masera, O. (2000). Community forest management in Mexico: carbon mitigation and biodiversity conservation through rural development. *Global Environmental Change, 10*, 259–272.

Kremen, C., Niles, J., Dalton, M., Daily, G., Ehrlich, P., Fay, P., *et al.* (2000). Economic incentives for rain forest conservation across scales. *Science, 288*, 1828–1832.

Latifovic, R., Zhu, Z., Cihlar, J., & Giri, C. (2003). *Land cover of North America 2000*, version 4. Natural Resources Canada, Canada Center for Remote Sensing, and EROS Data Center.

Lee, D. R., & Barrett, C. (Eds.) (2000). *Tradeoffs or synergies? Agricultural intensification, economic development and the environment*. Wallingford, UK: CAB International.

McKinney, M. L. (2001). Role of human population size in raising bird and mammal threat among nations. *Animal Conservation, 4*, 45–57.

McNeely, J. A., & Scherr, S. J. (2002). *Ecoagriculture: strategies to feed the world and save wild biodiversity*. Covelo, CA, Washington, DC: Island Press.
Mellor, J. W. (2002). *Poverty reduction and biodiversity conservation: The complex role for intensifying agriculture*. Washington, DC: World Wildlife Fund.
Mittermeier, R. A., Myers, N., Thomsen, J. B., da Fonseca, G. A. B., & Olivieri, S. (1998). Biodiversity hotspots and major tropical wilderness areas: approaches to setting conservation priorities. *Conservation Biology, 12*, 516–520.
Padberg, M. W. (1979). Covering, packing and the knapsack problems. *Annals of Discrete Mathematics, 4*, 265–287.
Patterson, B. D., Ceballos, G., Sechrest, W., Tognelli, M. F., Brooks, T., Luna, L., *et al.* (2003). *Digital distribution maps of the mammals of the Western Hemisphere* version 1.0. Arlington, VA: NatureServe.
Pimm, S. L., Ayres, M., Balmford, A., Branch, G., Brandon, K., Brooks, T., *et al.* (2001). Can we defy nature's end? *Science, 293*, 2207–2208.
Pressey, R. L. (1994). Ad hoc reservations—forward or backward steps in developing representative reserve systems. *Conservation Biology, 8*, 662–668.
Pressey, R. L. (1998). Algorithms, politics, and timber: an example of the role of science in a public, political negotiation process over new conservation areas in production forests. In R. T. Wills & R. J. Hobbs (Eds.), *Ecology for everyone: Communicating ecology to scientists, the public, and politicians* (pp. 73–87). Sydney: Surrey Beatty & Sons.
Pressey, R. L., & Cowling, R. M. (2001). Reserve selection algorithms and the real world. *Conservation Biology, 15*, 275–277.
Pressey, R. L., Humphries, C. J., Margules, C. R., Vanewright, R. I., & Williams, P. H. (1993). Beyond opportunism—key principles for systematic reserve selection. *Trends in Ecology and Evolution, 8*, 124–128.
Redford, K. H. (1992). The empty forest. *Bioscience, 42*, 412–422.
Redford, K., & Richter, B. (1999). Conservation of biodiversity in a world of use. *Conservation Biology, 13*, 1246–1256.
Ridgely, R. S., Gallnut, T. F., Brooks, T., McNicol, D. K., Mehlman, D. W., Young, B. E., *et al.* (2003). *Digital distribution maps of the birds of the Western Hemisphere* version 1.0. Arlington, VA: NatureServe.
Rodrigues, A. S. L., Cerdeira, J. O., & Gaston, K. J. (2000). Flexibility, efficiency, and accountability: adapting reserve selection algorithms to more complex conservation problems. *Ecography, 23*, 565–574.
Rodrigues, A. S. L. *et al.* (2004). Effectiveness of the global protected area network in representing species diversity. *Nature, 428*(8), 640–643.
Sánchez-Azofeifa, G. A., Daily, G. C., Pfaff, A. S. P., & Busch, C. (2003). Integrity and isolation of Costa Rica's national parks and biological reserves: examining the dynamics of land-cover change. *Biological Conservation, 109*, 123–135.
Sanderson, E. W., Jaiteh, M., Levy, M. A., Redford, K. H., Wannebo, A. V., & Woolmer, G. (2002). The human footprint and the last of the wild. *Bioscience, 52*, 891–904.
Sanderson, J., Alger, K., da Fonseca, G. A. B., Galindo-Leal, C., Inchausty, V. H., & Morrison, K. (2003). *Biodiversity conservation corridors: Planning, implementing and monitoring sustainable landscapes*. Washington, DC: Conservation International.
Sanderson, S. (1998). The new politics of protected areas. In K. Brandon, K. Redford, & S. Sanderson (Eds.), *Parks in peril* (pp. 441–454). Washington, DC: Island Press.
Scott, J. M., Davis, F. W., McGhie, R. G., Wright, R. G., Groves, C., & Estes, J. (2001). Nature reserves: do they capture the full range of America's biological diversity? *Ecological Applications, 11*(4), 999–1007.
Segura, G. (2000). Mexico's forest sector and policies: a general perspective. Prepared for "Constituting the commons: crafting sustainable commons in the new millennium", the eighth conference of the International Association for the Study of Common Property, Bloomington, IN, USA, 31 May.
Turner, I. M., & Corlett, R. T. (1996). The conservation value of small, isolated fragments of lowland tropical rain forest. *Trends in Ecology and Evolution, 11*(8), 330–333.
Underhill, L. G. (1994). Optimal and suboptimal reserve selection algorithms. *Biological Conservation, 70*, 85–87.
UNESCO (2002). *Biosphere reserves: special places for people and nature*. Paris: UNESCO.
United Nations (1999). *Long-range world population projections: based on the 1998 revision*. New York: United Nations Population Division.
United States Agency for International Development (2002). *Biodiversity and tropical forest conservation assessment* (pp. 118–119).
Vane Wright, R. I., Humphries, C. J., & Williams, P. H. (1991). What to protect—systematics and the agony of choice. *Biological Conservation, 55*, 235–254.
Velázquez, A., Mas, J. F., Palacio, J. L., & Bocco, G. (2002). Land cover mapping to obtain a current profile of deforestation in Mexico. *Unasylva, 210*, 8–13.
Velázquez, A., Mas, J. F., Díaz-Gallegos, J. R., Mayorga-Saucedo, R., Alcántara, P. C., Castro, R., *et al.* (2002). Patrones y tasas de cambio del uso del suelo en Méxicoc. *Gaceta Ecológica, 62*, 21–37.
Wells, M., & Brandon, K. (1992). *People and parks: linking protected area management with local communities*. Washington, D.C.: The World Bank.
Winters, P., Davis, B., & Corral, L. (2002). Assets, activities and income generation in rural Mexico: factoring in social and public capital. *Agricultural Economics, 27*, 139–156.