BIOTROPICA
THE JOURNAL OF TROPICAL BIOLOGY AND CONSERVATION

BIOTROPICA 40(3): 285–294 2008 10.1111/j.1744-7429.2007.00382.x

Patterns of Land Cover Change in and Around Madidi National Park, Bolivia

Jessica L. Forrest[1,2,6], **Eric W. Sanderson**[2], **Robert Wallace**[3], **Teddy Marcelo Siles Lazzo**[3], **Luis Humberto Gómez Cerveró**[3,4], and **Peter Coppolillo**[2,5]

[1]Department of Ecology, Evolution, and Environmental Biology and the Center for Environmental Research and Conservation (CERC), Columbia University, 10th Floor Schermerhorn Extension, 1200 Amsterdam Avenue, New York, New York 10027, U.S.A.

[2]Living Landscapes Program, Wildlife Conservation Society, 2300 Southern Blvd., Bronx, New York 10460, U.S.A.

[3]Greater Madidi Landscape Program, Wildlife Conservation Society, Casilla No. 3-35181, #133, Calle 11, Obrajes, La Paz, Bolivia

[4]Fundación Amigos de la Naturaleza, Km. 7 Carretera Antigua a Cochabamba, Casilla 2241, Santa Cruz, Bolivia

[5]Rungwa Ruaha Landscape Program, Wildlife Conservation Society, SLP 1654, Iringa, Tanzania

ABSTRACT

This study examines how human land uses and biophysical factors serve as predictors of land cover change in and around Madidi National Park in Bolivia. The Greater Madidi Landscape ranges over an elevational gradient from < 200 m in the Amazon basin to 6000 m in the high Andes, contains more than ten major ecosystem types, and several protected areas and sustainable use zones. In this study, Landsat Thematic Mapper satellite images collected over the study area at the beginning of the 1990s and then the 2000s were classified according to broad land cover types. Below elevations of 3000 m, the landscape experienced equal rates of deforestation and secondary forest increases of approximately 0.63 percent annually, resulting in no significant net change. Below elevations of 1000 m, however, we found an annual net loss in forest cover of 0.11 percent. Across the landscape, land cover change was most likely to occur near areas previously deforested, near roads and population centers, and at low elevations. We found net deforestation rates to be inversely related to strength of natural resource protection laws in protected areas and other jurisdictions. Results suggest little net change for the landscape as a whole, but that local scale changes may be significant, particularly near roads. Management policies favorable for biodiversity conservation in this landscape should limit the building of new roads and immigration to biologically sensitive areas and continue to support protected areas, which are achieving a positive result for forest conservation.

Abstract in Spanish is available at http://blackwell-synergy.com/loi/btp.

Key words: Conservation; deforestation; development; forest; land use; protected area; remote sensing.

Recent studies have shown that land cover change is rapidly reconfiguring tropical forest landscapes around the world (Skole & Tucker 1993, Laurence *et al.* 2001, Kinnaird *et al.* 2003, Zhang *et al.* 2005). Biophysical factors affecting rates and patterns of land cover change include elevation, slope, climate, and soil type (Etter & Villa 2000, Laurence *et al.* 2002); while human predictors include roads and highways, population size and density, and management area boundaries (Hayes *et al.* 2002, Müller & Zeller 2002, Sanchez-Azofeifa *et al.* 2002). Most studies show that rather than there being a single overriding factor, land cover change is multicausal and influenced by factors both near to and distant from the landscape (Geist & Lambin 2002).

Land cover change is a concern in tropical landscapes because these areas often have high levels of biodiversity and endemism, and because they maintain ecosystem services such as watershed protection, climate regulation, and carbon sequestration (Salati & Nobre 1991, Gentry 1995, Houghton *et al.* 2000, Myers *et al.* 2000, Laurence *et al.* 2002). The Greater Madidi Landscape, located in northwestern Bolivia and centered on Madidi National Park, is one such landscape of global conservation value (Painter *et al.* 2006). This landscape ranges from the high Andes to the Amazon basin over a vast area of 50,000 km^2 and across multiple jurisdictions. It hosts some of the world's highest levels of endemism and a number of globally endangered species, including spectacled bear (*Tremarctos ornatus*), vicuňa (*Vicugna vicugna*), Andean condor (*Vultur gryphus*), jaguar (*Panthera onca*), giant river otter (*Pteronura brasiliensis*), and peccary (*Tayassu pecari*) (Coppolillo *et al.* 2004, Painter *et al.* 2006).

Humans have played an integral role in the Madidi landscape since pre-Incan times, contributing to its structural complexity through their interactions with the land and their dependency on its natural resources (Silva *et al.* 2002). Current use of the landscape has evolved to represent diverse human interests including industrial resource extraction, agriculture, tourism, communal management and use by indigenous peoples, and conservation of biodiversity (Painter *et al.* 2006). A few recent studies measured land cover change in localized portions of this landscape. Tucker and Townshend (2000) describe a total annual forest loss of 311 ha (1.9 percent) in the eastern lowland portion of the study area and a 1 ha (0 percent) loss in the western lowland portion over the years 1985–1986, from an analysis of satellite imagery (also Townshend *et al.* 1995). Locklin and Haack (2003) studied a subset of the region associated with an approximately 90 km stretch of road between the villages of 30 Agosto and El Tigre, estimating that 3448 ha was lost between 1987 and 2000. A similar study examining change along the Yucumo-Rurrenabaque portion of the road adjacent to Pilón Lajas National Park and Biosphere Reserve estimated a 3208 ha net decrease in forest cover along the approximately 110 km stretch between 1987

Received 15 November 2006; revision accepted 15 August 2007.
[6]Corresponding author; e-mail: jessica.forrest@wwfus.org

and 2004, although almost half of the observed change occurred in the final 3 yr of the study period (WCS Greater Madidi Landscape Conservation Program 2005, unpublished data.).

In this study, we expand upon these localized studies by examining land cover change relative to biophysical and human factors throughout the low and mid-elevation reaches of the Bolivian portion of the Greater Madidi Landscape from 1990 to 2000. Specifically, we quantify the landscape-wide rate of change of both deforestation and secondary growth of humid forest types, while also examining how proximity to roads, population centers, and previous deforestation; elevation; and different protected area designations influence the patterns of land cover change seen in this landscape. We discuss the results with particular attention to human drivers of change in the landscape.

METHODS

STUDY AREA.—The study area covers the primarily forested part of the Greater Madidi Landscape on the eastern flank of the Andes in Northwest Bolivia, with a total area of approximately 34,000 km^2 and a center point at 13°52′42.9″ S, 56°11′26.7″ W. It is bounded by the Peruvian border to the west, the 3000 m contour to the south, and the satellite image extent to the south, east, and north (Fig. 1). The study area includes several managed area types, including Madidi National Park, the Madidi and Apolobamba Integrated Management Areas, the Pilón Lajas Biosphere Reserve and Indigenous Territory, the Tacana and San José de Uchupiamonas indigenous territories, forestry concessions, and 'unmanaged areas' not subject to unique management designations. Although different forest types—including cloud, humid montane, dry, and lowland—dominate most of the study area, significant grassland and savanna areas also exist. These include the mid-elevation Apolo grasslands, which make up an area of *ca* 1300 km^2 in the center of the study area; and low elevation mosaics of natural savanna, chaparral, wetlands, lakes, and large rivers including the Beni and Tuichi rivers. Madidi also boasts the largest remaining dry tropical forest area in South America: with an area > 550 km^2 in the mid-elevation reaches of the landscape, this ecosystem harbors the most diverse abundance of dry forest species in the Neotropics (Kessler *et al.* 1998). In this study, we measure change in the extent of humid forest and nonforest cover types, but exclude change in dry forest from the analysis.

DATA SELECTION.—Two adjacent pairs of satellite images from the Landsat Thematic Mapper (TM) and Landsat Enhanced Thematic Mapper (ETM+) platforms were selected to assess land cover change in the study area. Criteria used to select images included collection dates (*ca* 1990 and 2000), minimal cloud cover, availability at little to no cost, and when possible, seasonality. Images were acquired free of charge from the Global Land Cover Facility website (http://glcf.umiacs.umd.edu).

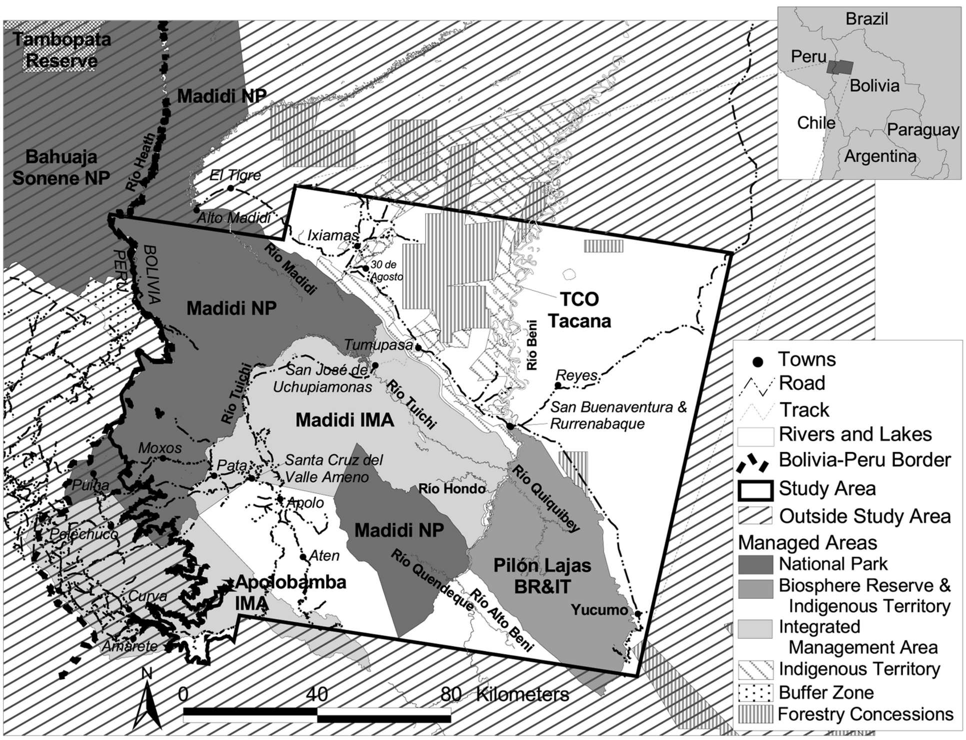

FIGURE 1. The study area. Management units include Madidi National Park (NP), Madidi Integrated Management Area (IMA), Pilón Lajas Biosphere Reserve and Indigenous Territory (BR&IT), Tacana Indigenous Territory (TCO), forestry concessions, and unprotected areas not subject to specific natural resource management strategies.

The selected images were dated 2 August 1993 (TM) and 29 June 2001 (ETM+) over the eastern Madidi landscape (Path 1/Row 70) which includes 81 percent of the study area; and 21 September 1990 and 30 April 2000 over the western 19 percent of the landscape (Path 2/Row 70). The time series pair over eastern Madidi was collected 7 yr and 11 mo apart in the dry season. It incorporates the entire Amazonian portion of the study area, as well as the majority of montane areas. The time series pair over western Madidi (Path 2/Row 70) was collected in the dry season in 1990 and the transition between the wet and dry season in 2000. The western Madidi time series consists entirely of mid- and high elevation areas, and is predominantly represented by humid and dry tropical forest cover types. The time series pairs are variably affected by seasonal and even monthly differences in ecosystem phenology based on the collection date and the ecosystem type (Kaláscka *et al.* 2005, Myeni *et al.* 2007); with the western time series most significantly affected. These inherent challenges in the image data were addressed during the image classification process.

DATA PREPARATION AND CLASSIFICATION OF IMAGE PAIRS.—Using ERDAS Imagine software (version 8.4, from ERDAS, Atlanta Georgia), the 1993 image collected over eastern Madidi was coregistered to its 2001 time-series pair using 20 ground control points (GCPs) distributed evenly throughout the image. The coregistration process was repeated for the images in the western portion of the study area using 15 GCPs. Following geometric correction and resampling, the total root mean square (RMS) error for both image pairs was calculated to be < 28.5 m, or one pixel length.

To minimize classification error from haze and elevation, each image was divided into two to three subset images for further processing, based on similar atmospheric and spectral characteristics in the subset images (Lillesand & Kiefer 2000). For example, all flat areas < 500 m in the Amazon basin were subset into one image, while the second image subset consisted of the montane areas in the same original image. To maximize contrast between forest and nonforest cover types, we displayed subset images with bands 4, 5, and 3 in red, green, and blue respectively, and applied a Gaussian stretch. Next, we undertook a separate supervised classification procedure for each subset image to represent four major cover classes (humid forest; dry forest; grassland, savannah, and cropland; and water) and three no data classes (cloud, cloud shadow, and mountain shadow). We chose this process of interpreting and classifying images uniquely in order to control for differences between image dates caused by sensor types (TM versus ETM+) and seasonal variations in the spectral signatures of vegetation. Training sets were based on vegetation ground data collected by the Wildlife Conservation Society (WCS) in Bolivia. Where ground data were not available, training sets were developed based on recommendations from locally knowledgeable experts.

Following the preparation of training sets, a maximum-likelihood supervised classification was run separately on each of the 6-band subset images. Following the first iteration of each classification, the output image was examined, and poorly constructed training sets were identified by running a contingency matrix. Training sets were edited and the maximum-likelihood classification was rerun. This process was repeated 10–15 times per subset image until the cover classes produced by the supervised classification were considered satisfactory based on visual comparison of the classification to the raw satellite image.

POSTCLASSIFICATION DATA PROCESSING.—A postclassification smoothing procedure based on a nearest-neighbor algorithm was applied to each classified image to reduce the salt-and-pepper appearance of the data. The smoothed images have a minimum mapping unit of 11 pixels, or < 1 ha.

Using Imagine software, the classified images at Path 1/Row 70 were projected to the *Universal Transverse Mercator (UTM) WGS 1984 Zone 19 South* projection to match the projection of the adjacent images. The maximum error tolerance during resampling was set at 0.1 pixel. The 1990 and 1993 images were then mosaicked to form one large image (referred to hereafter as the 1990 mosaic), while the 2000 and 2001 images were mosaicked to form a second large image (referred to as the 2000 mosaic). The mosaics from the 2 yr were then compared to ensure that permanent features, such as roads and montane rivers, from the two mosaics overlay each other without error.

A series of postclassification procedures were subsequently undertaken in ArcView 3.2 GIS (Environmental Systems Research Institute, Redlands, California) to eliminate misclassified land cover and cloud types, which occurred when the reflectance patterns of one cover type resembled that of another. Specifically, cloud edge areas misclassified as grassland were identified by delineating polygons around the problem areas and then reclassified to the correct class. A similar GIS procedure was used to delineate the mid-elevation dry forest area, which varied drastically in spectral characteristics between the two image collection dates due to seasonal change. The area of dry forest was identified as the large contiguous area of change between the two images, north of the Saucemayo River and extending up to the unchanged humid forest area approximately 60 km northwest of Apolo. Since the objective of this study was to document changes in humid forest, this delineation procedure was regarded as acceptable for identifying and correcting dry forest areas subject to seasonal differences, without affecting rates of change for humid forest.

ACCURACY ASSESSMENT.—Land cover points used to assess the accuracy of the 2000 land cover mosaic were gathered from two main sources, on-the-ground surveys and comparison at random locations with a land cover map of South America (Eva *et al.* 2004). The ground survey data consisted of 221 points collected between 1998 and 2004, distributed across elevations, and located in both eastern and western Madidi (WCS Greater Madidi Landscape Conservation Program 2007, unpublished data.). Since the points initially described more detailed habitat types than are relevant to this study, each point was reassigned to an appropriate broad land cover class consistent with those used for the change analysis. For example, cloud, humid, and dry forests were all assigned to the all-inclusive 'forest' class; savannah, pasture, agriculture, and scrub were all assigned to the 'nonforest' class. To eliminate bias due to spatial autocorrelation, the points were subset to only one point per 1-ha area, the minimum mapping unit of the study. Since all but eight of the ground-gathered vegetation points identified forest cover, we complemented these with 46 additional points that were randomly generated in nonforest cover types across the landscape as determined by a regional land cover classification for South America, the GLC2000-South America (Eva *et al.* 2004). This map was generated from three low-resolution sensors, the Along Track Scanning Radiometer (ATSR-2); SPOT VEGETATION (VGT); and The Defense Meteorological Satellite Program (DMSP) Operational Linescan System (OLS). To compare the land cover map prepared for this study to the GLC2000-South America, we resampled the 2000 Mosaic to 1 km^2 to have a comparable resolution with the GLC-2000, and then assigned each random point with the land cover class at that grid cell. From the full set of accuracy assessment points, a confusion matrix was generated, and rates of overall accuracy and accuracy by land cover class were calculated as per methods described by Congalton (1991). Unfortunately, ground data were not available to assess the accuracy of the 1990 mosaic. We assume, however, that the classification accuracy of the 1990 mosaic is similar to that of the 2000 mosaic since the same methods were used to classify both mosaics.

PREPARATION AND ANALYSIS OF CHANGE DATA.—The areas and locations of humid forest loss, increase in secondary humid forest, and no change were derived from the classified 1990 and 2000 mosaics using the Map Calculator available in ArcView 3.2 Spatial Analyst. Areas that were classified as water, no data (due to cloud or shadow), or dry forest in 1990 and/or 2000 were excluded from the change analysis and marked as 'no data.' Hereafter, we refer to the map resulting from this calculation as the Change Mosaic. Next, land cover change was assessed with respect to distance to areas previously deforested, roads and population centers, and elevation.

To examine the relationship between land management regimes and land cover change from human disturbance, we analyzed change in the extent of humid forest within the following jurisdictions: Madidi National Park, Madidi Integrated Management Area (IMA), the Tierras Communitarias de Origen Tacana (TCO Tacana), all forestry concessions, and unprotected areas. We limited this analysis to areas clearly subject to higher impact from humans: within 3 km of roads and population centers throughout the study area, and excluding frequently flooded areas such as the flood plain of the Beni river where natural change is extremely high. To detect possible encroachment into the national park, we generated buffers at increasing distances within the boundaries of Madidi National Park and assessed the amount of change in each distance zone.

RESULTS

ACCURACY ASSESSMENT RESULTS.—The accuracy assessment estimates that the 2000 mosaic has an overall accuracy of 90 percent. The individual classes relevant to this study, humid forest, and nonforest cover types, also meet accuracy thresholds identified by Thomlinson *et al.* 1999 (Table 1).

TABLE 1. *Confusion matrix for the 2000/2001 land cover map.*

	Actual class (Groundtruth data)			User's
Predicted class (2000/01 mosaic)	Forest	Nonforest	Total #	Accuracy (%)
Forest	196	10	206	95
Nonforest	17	44	61	72
Total #	213	54	267	
Producer's accuracy (%)	92	81		Overall accuracy: 90%

BROAD-SCALE TRENDS.—Over the entire Madidi landscape, we saw no net change in forest cover annually from 1990 to 2000, with estimated rates of deforestation and secondary forest increases each equal to 0.63 percent per year. Below elevations of 1000 m, however, we found a net decrease in forest cover of 0.11 percent annually. The rates of humid forest loss and increase at these low elevations were five or six times the net rate of change, implying this is a landscape far more dynamic than net rates suggest (Table 2).

We found that localized areas of land cover change were more significant than change across the study area as a whole. The rates of change appeared to be highest in the Amazon Basin in areas frequently flooded at the interface between natural savanna and humid tropical forest, along the Beni River, at expanding agricultural plots along the Yucumo-Rurrenabaque-Ixiamas-El Tigre road (referred to hereafter as the Yucumo-Ixiamas road), and emanating outwards from the Apolo grasslands. Along the Yucumo-Ixiamas road (the most visible sign by satellite of human habitation in the area) and near the Apolo grasslands, we observed a net deforestation trend over the study interval (Fig. 2).

ROADS AND POPULATION CENTERS AS PREDICTORS OF CHANGE.—The rate of forest loss increased with increasing proximity to roads and population centers. In general, forest cover showed an overall net decrease within 4 km of roads and population centers, but stabilized at a minimal rate of net change at distances > 4 km from these developments (Fig. 3).

PREVIOUS DEFORESTATION AS A PREDICTOR OF CHANGE.—Areas already deforested by the year 1993 were also examined as predictors of change (Fig. 4). This analysis was only done for the eastern image (Path 1/Row 70) due to better information on anthropogenic change than was available for the western part of the study area (Beck *et al.* 2002, Silva *et al.* 2002). We found distance to previous deforestation to be the most important predictor of net forest loss, which occurred at a rate of 1.9 percent annually within 1 km of previous deforestation. This rate of deforestation gradually decreased as distance from the source increases, until it reached an equilibrium rate of minimal net change between 6 and 10 km.

MANAGEMENT REGIME AS A PREDICTOR OF CHANGE.—We found that the rate of land cover change due to human activity varied

TABLE 2. *Area and rates of land cover change in the Madidi landscape.*

Image	Original extent of forest studied (km^2)	Net Percent Change/yr	Change Class	Km^2 Change/yr	Percent Change/yr
Below 3000 m					
Madidi East[a]	18,350	−0.042	Secondary forest	122	0.67
			Forest loss	−130	−0.71
Madidi West[b]	4792	+0.21	Secondary forest	24	0.50
			Forest loss	−14	−0.30
Change Mosaic[c]	23,141	0.0	Secondary forest	146	0.63
			Forest loss	−144	−0.63
Below 1000 m					
Madidi East[a]	14,904	−0.12	Secondary forest	82	0.55
			Forest loss	−100	−0.67
Madidi West[b]	1349	+0.037	Secondary forest	2	0.16
			Forest loss	−2	−0.13
Change Mosaic[c]	16,253	−0.11	Secondary forest	84	0.52
			Forest loss	−102	−0.64

[a]Image location: Path 1/Row 70, Interval: 1993–2001.

[b]Image location: Path 2/Row 70, not including area of overlap with images at Path 1/Row 70, Interval: 1990–2000.

[c]Mosaic of Madidi East and West.

significantly with land management regime (see Fig. 5). The forestry concessions showed the highest rate of deforestation compared with other jurisdictions, due to a large area cleared in one concession on the Yucumo-Ixiamas road. Like the forestry concessions, unprotected areas not subject to special natural resource management designations exhibited a net decrease in forest cover. The IMA and the national park, both managed by the Bolivian System of Protected Areas (SERNAP), showed net increases in forest cover over the study period. When land cover change in the national park was analyzed by distance to park edge; however, we observed a net decrease in forest cover within 4 km of the park boundary, and a net increase toward the interior (Fig. 6). In the indigenous territory, where natural resources are managed locally by the Tacana indigenous community, we find a minimal net loss in forest cover. These results suggest that higher levels of natural resource protection in protected areas correlate with lower net rates of forest loss, but that rates of change are also not uniform within the protected areas.

DISCUSSION

SOURCES OF ERROR.—Land cover maps and change assessments are subject to error from a variety of sources (Foody 2002). Variations in elevation, atmospheric conditions, seasonality, and shadow within images often lead to confusion between classes. The process of classifying mixed pixels and areas of degradation into discrete classes results in a simplified model of land cover, underestimating forest cover in some pixels while overestimating it in others (Congalton 1988, Steele *et al.* 1998, Foody 2002). In land cover change maps, error tends to be compounded since inaccuracies in individual classifications are confused with change (Khorram 1999).

In this study, we minimized inaccuracy through our methodology in several ways. These included selecting images from the same season when possible; minimizing error in coregistration between time series pairs by ensuring overall RMS error was less than one pixel; and classifying images in the time series separately to account for variation in spectral signatures caused by different sensor types and season of collection. We also used postclassification procedures to address confusion between classes; and eliminate areas of speckle that would otherwise appear as change. As a result, we obtained an overall accuracy of 90 percent, which is comparable to land cover analyses in other tropical landscapes (Sanchez-Azofeifa *et al.* 2003, Gaveau *et al.* 2007) and meets recommended accuracy standards for land cover maps derived from satellite imagery (Thomlinson *et al.* 1999, Foody 2002).

BROAD-SCALE PATTERNS.—The Madidi landscape has experienced relatively low rates of land cover change during the 8- to 10-yr study interval when compared with other tropical forest landscapes worldwide (Skole & Tucker 1993, Seddon *et al.* 2000, Laurence *et al.* 2001, Zarin *et al.* 2001, Hayes *et al.* 2002, Kinnaird *et al.* 2003). Indeed, the landscape as a whole showed no net change in forest cover, although we observed a deforestation trend at the lowest elevations. When we compared the current rate of change in the eastern lowland portion (< 1000 m) of the landscape to earlier estimates of change across the same extent, we found that the rate of forest loss was significantly lower than estimates produced by Tucker and Townshend for 1985–1986. Compared with the Tierras Bajas region of Santa Cruz, Bolivia (Tucker & Townshend 2000, Steininger *et al.* 2001a), Madidi has experienced far lower rates of deforestation, indicating that the rate of forest cover change within Bolivia is far from uniform (Steininger *et al.* 2001b).

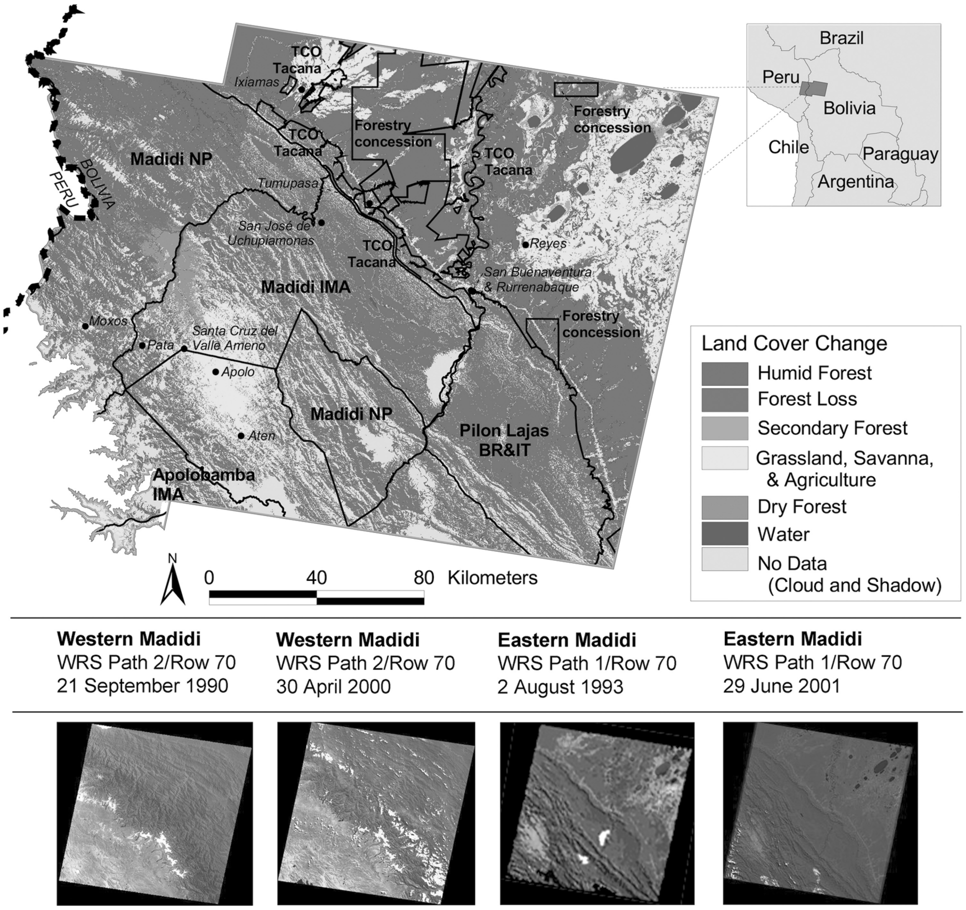

FIGURE 2. Map of land cover change in Madidi and original satellite images. Here, humid forest = humid forest in both 1990 and 2000; forest loss = humid forest in 1990 and grassland, savanna, agriculture, or other nonforest cover in 2000; secondary forest = nonforest cover in 1990 and humid forest in 2000; dry forest = dry forest in 1990 and/or 2000; water = water in 1990 and/or 2000; no data = cloud or shadow in 1990 and/or 2000.

The disparity in rates between the two landscapes is probably due to differences in the underlying causes of deforestation—the Tierras Bajas are being cleared primarily for industrial-scale agriculture aimed at producing soybeans and other export crops (Steininger *et al.* 2001a), while much of the clearing in Madidi is associated with small-scale agriculture near roads.

The most important cause of deforestation in the Madidi landscape over the study interval was agriculture and development along the Yucumo-Ixiamas road, which is a pattern consistent with other tropical landscapes in Latin America (Geist & Lambin 2002). The Yucumo-Ixiamas road was constructed from the east to Rurrenabaque in 1989 and then extended to Ixiamas in 1991 (Silva *et al.* 2002). Originally built by a logging company to serve its mill in Alto Madidi, the road is being settled by a variety of people including local Aymara and Quechua people from the Bolivian highlands and tropical Alto Beni, and immigrants from other parts of Bolivia, Argentina, and even Russia. Settlement along the road is government-directed in some areas like El Tigre and spontaneous in others (Locklin & Haack 2003). The Yucumo-Ixiamas road represents a linear source of fragmentation to the landscape, with possible consequences for ecological processes such as pollination and species dispersal. It may also serve as a conduit for invasive species (Janzen 1983, Lovejoy *et al.* 1986, Skole & Tucker 1993, Forys 2002, Laurence *et al.* 2002).

Another source of human-caused change to the study area, although occurring at a far smaller spatial and temporal scale than change along the Yucumo-Ixiamas road, results from the traditional small-scale subsistence agriculture practiced by indigenous communities such as the Tacana. As such, people clear approximately 1 to 2 ha of land for 2–3 yrs before allowing the land to revert to forest. This practice is not necessarily a threat to biodiversity; indeed, when confined to a small scale in both space and time, subsistence agriculture can be consistent with the long-term maintenance of a heterogeneous ecosystem, with little impact on the landscape as a whole (Andrade & Rubio 1994, Kattan & Murcia 2002). Some studies have even shown that moderate levels of disturbance and traditional management may in fact enhance species diversity in forested tropical landscapes (Connell 1978, Kessler 2001, Pinedo-Vasquez *et al.* 2002).

While the low elevation portion of the landscape had the most significant source of human disturbance observable by satellite due to the Yucumo-Ixiamas road, it is notable that the lowest elevations

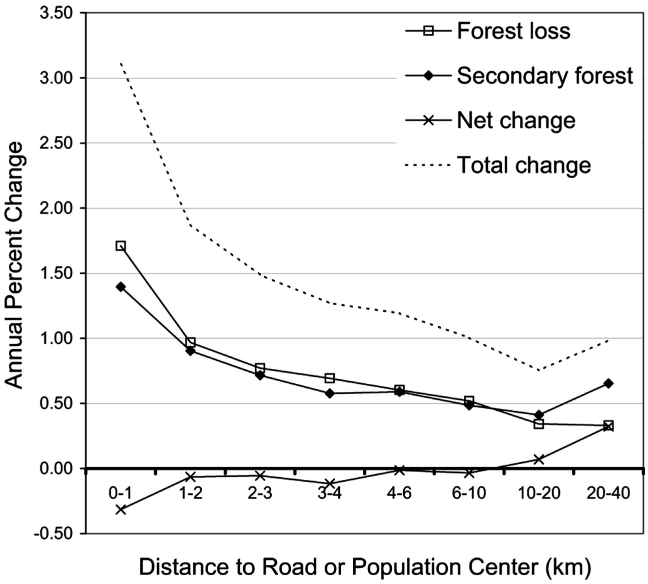

FIGURE 3. Rate of land cover change with respect to distance to roads and population centers. Total land cover change increases with increasing proximity to roads and population centers, and a net deforestation trend exists within 4 km of these features.

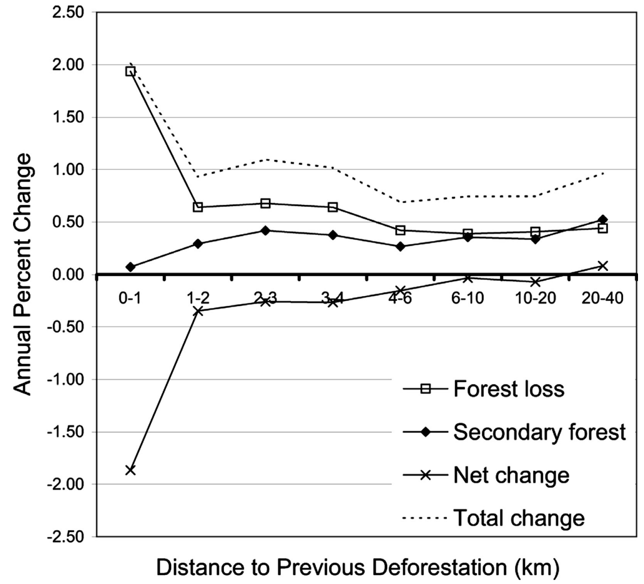

FIGURE 4. Rate of land cover change with respect to distance to previous deforestation. Total land cover change increases with increasing proximity to areas previously deforested, and a strong deforestation trend exists within 1 km of these features. Net change in forest cover approaches 0 at distances of 6 to 10 km from areas previously deforested.

in the study area were also naturally far more dynamic than the higher elevation areas. This is a result of variable amounts of flooding and drought that occur in the Amazon basin each year, causing long-term shifts in the location of open water sources such as oxbow lakes and vegetation observed in the same season over the 8-yr period.

The variation in the rate of human-caused disturbance between low and high elevation areas remains a concern since the low areas provide habitat to a different community of species from the high elevation areas. In addition, as human expansion continues in the lowlands, it may limit the ability of the ecosystem to respond to natural disturbances. For this reason, natural resource management strategies should ensure that the spatial and temporal scale of human disturbances are synchronized as much as possible to the scale of natural disturbance regimes (Pickett *et al.* 1997).

MANAGEMENT REGIME AS A PREDICTOR OF LAND COVER CHANGE.— The results of this study demonstrate that the strength and type of natural resource protection laws appear to have a clear relationship to net rates of forest cover change in the Madidi landscape. As such, areas managed by SERNAP show a net increase in forest cover; the indigenous territory shows a rate of net change close to null; and all other areas show a significant net decrease.

The positive result for the national park and IMA can be attributed to a variety of factors, ranging from favorable conditions in the national park and IMA at the time of their formal designation, to effective park management. Conditions considered favorable for conservation include relatively low human population sizes, minimal infrastructure such as roads and buildings, and relatively steep terrain in the national park and IMA, which discourage human access. In addition to these favorable conditions, it appears that protected status has also benefited biodiversity in the area. Before 1995, wildlife populations along the Hondo and Tuichi rivers had been severely reduced, and forests degraded by a decade of selective logging. Wallace *et al.* (2003), however, noted a recovery

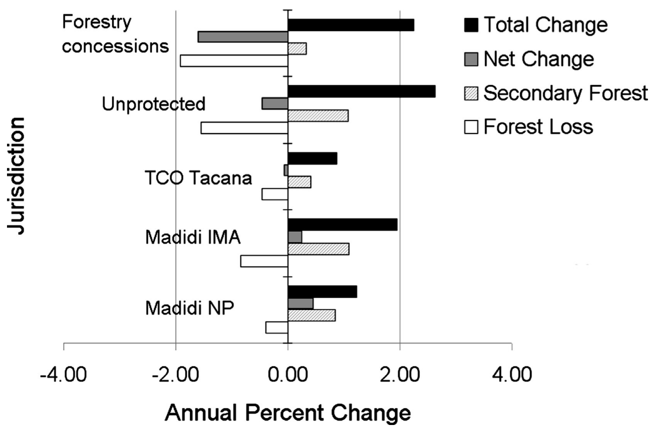

FIGURE 5. Rate of land cover change with respect to jurisdiction. Net change in forest cover is strongly related to the strength of natural resource protection laws, with the national park and integrated management area showing positive change in forest cover; the indigenous territory showing minimal net change, and the unprotected areas and forestry concessions showing significant loss of forest cover during the study period.

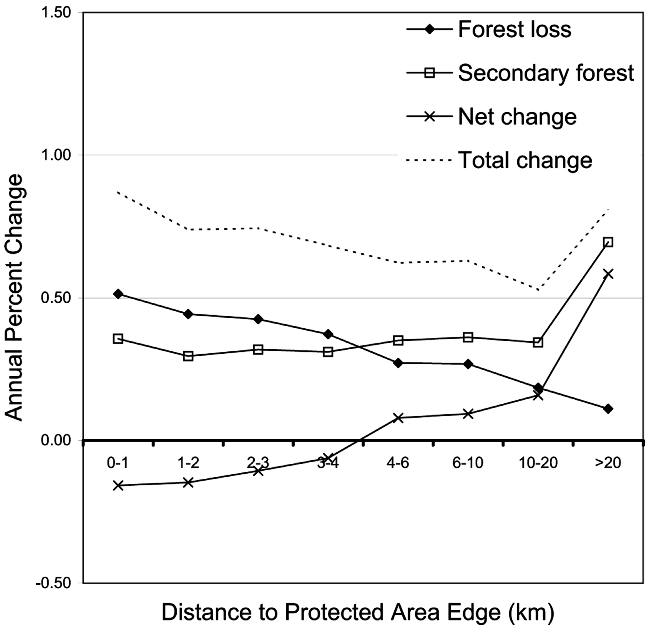

FIGURE 6. Rate of land cover change by distance to Madidi National Park boundary. Within Madidi National Park, areas of net forest loss tend to be concentrated within 4 km of the boundary of the protected area, while net increases in forest cover are concentrated toward its interior.

in these populations since park status was established and logging was banned. More recently, Madidi had to suspend most activities, including control and vigilance efforts, during a brief administrative crisis in SERNAP in 2006. This crisis resulted in several illegal encroachments in the protected area, suggesting that the protected area administration had previously provided effective management (R. Wallace, pers. comm.).

While the above results suggest a generally positive result for forest conservation in Madidi national park, nonuniform patterns of change within the protected area provide cause for concern. In this study, we noted a slight net deforestation trend within distances of 4 km inside the park boundary. This trend is caused by human-caused degradation and encroachment from the Apolo grasslands into the national park. We think the overall rate of deforestation near park boundaries is overestimated, however, due to an assumed overestimate of deforestation in the southwest corner of the protected area at the Peru border. While the pattern of change within the park boundaries is indeed a concern, we find that the rate of deforestation is lower than rates of change near roads and areas previously deforested.

Other landscapes that show generally positive trends in forest cover within protected areas include Celaque National Park in Honduras and the central highlands of Vietnam (Müller & Zeller 2002, Munroe *et al.* 2002). The authors of these two studies suggest that protected area status is only one of several factors affecting forest regrowth rates, which range from voluntary abandonment of swidden agricultural fields for intensive permanent agricultural fields in both Honduras and Vietnam (Müller & Zeller 2002, Munroe *et al.* 2002), to a government colonization policy in Vietnam (Müller & Zeller 2002). Gaveau *et al.* (2007), on the other hand, found that protected area designation led to a significant decrease in large-scale mechanized logging, but did not reduce deforestation due to agricultural encroachment in Sumatra's Bukit Barisan Selatan National Park in Indonesia. The authors attributed this phenomenon to a rapidly increasing rural population resulting from spontaneous migrations in the late 1990s. In Madidi, the main locations of park encroachment during the study period involve 500 ha of cleared forest in the northern section of the Pilón Lajas Biosphere Reserve and Indigenous Territory, located near the Yucumo-Ixiamas road; and about 880 ha of localized deforestation within Madidi National Park, northwest of the Apolo grasslands. Pressures to Pilón Lajas result from colonization along the road by new immigrants to the area, while the pressures to Madidi come from local communities. Outside the limits of the study area, but still within the Greater Madidi Landscape, colonization is also occurring along the boundary of the Bahuaja-Sonene and Tambopata protected areas on the imminent Southern Peru Inter-Oceanic Highway.

In addition to localized areas of deforestation within the protected areas, high rates of deforestation outside the northern boundary of Madidi and Pilón Lajas raise concerns about the future isolation of these areas, as highlighted in Costa Rica (Sanchez-Azofeifa *et al.* 2002, 2003). While the effects of protected area isolation in the Madidi landscape might be muted somewhat by the scale of the area under protection, we find that the effects of isolation could nonetheless be severe for the widest ranging landscape species, including white-lipped peccary, jaguar, and Andean bear (Sanderson *et al.* 2002, Coppolillo *et al.* 2004, Painter *et al.* 2006). In addition, deforestation trends outside the protected areas could leave lowland species particularly vulnerable.

CONCLUSION

This study presents a generally optimistic view for conservation of the Greater Madidi Landscape, in which rates of forest loss over the extent of the landscape are relatively low, and net change in forest extent is close to null. Localized areas of forest loss remain a concern, however, and could have broader scale impacts. Such areas of concern exist along the Yucumo-Ixiamas road and near other areas already deforested. Results suggest that natural resource management policies favorable for conservation should limit the building of new roads in biologically sensitive areas, minimize the extension of agricultural plots surrounding existing roads, and focus in particular on maintaining the low elevation biodiversity in the study area. Results also suggest that protected areas are achieving a generally positive result by conserving forest habitats in this landscape, but that nonuniform change within protected areas is a cause for concern. Since this study does not provide insight about subtle forest degradation, or the status of plant and animal communities that are also important indicators of ecosystem integrity, the results of this study should be used in conjunction with on-the-ground studies for a more holistic understanding of conservation issues affecting the Greater Madidi Landscape. The results reported here help to inform our understanding of the potential impacts that transportation, mining, and agricultural developments can have on surrounding ecosystems, with possible applications for planning and natural resources policy beyond Madidi.

ACKNOWLEDGMENTS

We would like to thank L. Painter and B. Ríos of WCS Bolivia, R. Sevillanos of SERNAP, and CARE-Bolivia for their help with fieldwork and extensive insights specific this study; G. Woolmer and M. Steininger for their technical guidance; S. Bergen for his review of the methodology; M. Pinedo-Vasquez for sharing his knowledge on comparative forested systems in Latin America; and the reviewers at Biotropica for their helpful comments on the manuscript. We thank the Wildlife Conservation Society Living Landscapes and Latin America Programs and the Center for Environmental Research and Conservation (CERC) for enabling this study, the Prospect Hill Foundation for support for landscape ecology at WCS, and the ESRI Conservation Program for software support. We also thank the NASA New Investigator Program (Grant NNG04GP73G) for support to E. W. S.

LITERATURE CITED

ANDRADE, G. I., AND H. RUBIO. 1994. Sustainable use of the tropical rain forest: Evidence from the avifauna in a shifting-cultivation habitat mosaic in the Colombian Amazon. Conserv. Biol. 8: 545–554.

BECK, S. G., E. E. GARCÍA, AND F. R. ZENTENO. 2002. Diagnostico de Flora y Vegetación del Parque Nacional y Area Natural de Manejo Integrado Madidi. Wildlife Conservation Society-La Paz and CARE-Bolivia. Funded by the European Community. La Paz, Bolivia.

CONGALTON, R. G. 1988. Using spatial autocorrelation analysis to explore the errors in maps generated from remotely sensed data. Photogrammetric Eng. Remote Sensing 54: 587–592.

CONGALTON, R. G. 1991. A review of assessing the accuracy of classifications of remotely sensed data. Remote Sens. Environ. 37: 35–46.

CONNELL, J. H. 1978. Diversity in tropical rain forests and coral reefs. Science 199: 1302–1310.

COPPOLILLO, P., H. GOMEZ, F. MAISELS, AND R. WALLACE. 2004. Selection criteria for suites of landscape species as a basis for site-based conservation. Biol. Conserv. 115: 419–430.

ETTER, A., AND L. A. VILLA. 2000. Andean forests and farming systems in part of the eastern Cordillera (Colombia). Mountain Res. Develop. 20: 235–245.

EVA, H. D., A. S. BELWARD, E. E. DE MIRANDA, C. M. DI BELLA, V. GOND, O. HUBER, S. JONES, M. SGRENZAROLI, AND S. FRITZ. 2004. A land cover map for South America. Global Change Biol. 10: 731–744.

FOODY, G. M. 2002. Status of land cover classification accuracy assessment. Remote Sens. Environ 80: 185–201.

FORYS, E. A., C. R. ALLEN, AND D. P. WOJCIK. 2002. Influence of the proximity and amount of human development and roads on the occurrence of the red imported fire ant in the lower Florida Keys. Biol. Conserv. 108: 27–33.

GAVEAU, D. L. A., H. WANDANO, AND F. SETIABUDI. 2007. Three decades of deforestation in southwest Sumatra: Have protected areas halted forest loss and logging, and promoted re-growth? Bio. Conserv. 134: 495–504.

GEIST, H. J., AND E. F. LAMBIN. 2002. Proximate causes and underlying driving forces of tropical deforestation. BioScience 52: 143–150.

GENTRY, A. H. 1995. Patterns of diversity and floristic composition in Neotropical montane forests. *In* S. P. Churchill, H. Balslev, E. Forero and J. L. Luteyn (Eds.). Biodiversity and conservation of Neotropical Montane Forests, pp. 103–126. The New York Botanical Garden, New York, New York.

HAYES, D. J., S. A. SADER, AND N. B. SCHWARTZ. 2002. Analyzing a forest conversion history database to explore the spatial and temporal characteristics of land cover change in Guatemala's Maya Biosphere Reserve. Landscape Ecol. 17: 299–314.

HOUGHTON, R. A., D. L. SKOLE, C. A. NOBRE, J. L. HACKLER, K. T. LAWRENCE, AND W. H. CHOMENTOWSKI. 2000. Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon. Nature 403: 301–305.

JANZEN, D. H. 1983. No park is an island: Increase in interference from outside as park size decreases. Oikos 41: 402–410.

KALÁCSKA, M. E. R., G. A. SÁNCHEZ-AZOFEIFA, J. C. CALVO-ALVARADO, B. RIVARD, AND M. QUESADA 2005. Effects of season and successional stage on leaf area index and spectral vegetation indices in three Mesoamerican tropical dry forests. Biotropica 37: 486–496.

KATTAN, G. H., AND C. MURCIA. 2002. A review and synthesis of conceptual frameworks for the study of forest fragmentation. *In* G. A. Bradshaw, P. A. Marquet, and K. L. Ronnenburg (Eds.). How landscapes change: Human disturbance and ecosystem fragmentation in the Americas, pp. 183–200. Springer, New York, New York.

KESSLER, M. 2001. Maximum plant-community endemism at intermediate intensities of anthropogenic disturbance in Bolivian montane forests. Conserv. Biol. 15: 634–641.

KESSLER, M., K. BACH, N. HELME, S. BECK, AND J. GONZALES. 1998. Floristic diversity of Andean dry forests in Bolivia—an overview. Proceedings of the Symposium der A.F.W. Schimoer-Stiftung.

KHORRAM, S. 1999. Accuracy assessment of remote sensing-derived change detection. American Society for Photogrammetry and Remote Sensing, Bethesda, Maryland.

KINNAIRD, M. F., E. W. SANDERSON, T. G. O'BRIEN, H. T. WIBISONO, AND G. WOOLMER. 2003. Deforestation trends in a tropical landscape and implications for endangered large mammals. Conserv. Biol. 17: 245–257.

LAURANCE, W. F., A. K. M. ALBERNAZ, AND C. DA COSTA. 2001. Is deforestation accelerating in the Brazilian Amazon? Environ. Conserv. 28: 305–311.

LAURENCE, W. F., K. M. ALBERNAZ, G. SCHROTH, P. M. FEARNSIDE, S. BERGEN, E. M. VENTICINQUE, AND C. DA COSTA. 2002. Predictors of deforestation in the Brazilian Amazon. J. Biogeogr. 29: 737–748.

LILLESAND, T. M., AND R. W. KIEFER. 2000. Remote sensing and image interpretation. John Wiley and Sons, New York, New York.

LOCKLIN, C. C., AND B. HAACK. 2003. Roadside measurement of deforestation in the Amazon area of Bolivia. Environ. Manage. 31: 774–783.

LOVEJOY, T. E., R. O. BIERREGAARD, A. RYLANDS, J. MALCOLM, C. QUINTELA, L. HARPER, K. BROWN, A. POWELL, G. POWELL, H. SCHUBART, AND M. HAYS. 1986. Edge and other effects of isolation on Amazon forest fragments. *In* M. E. Soulé (Ed.). Conservation biology: The science of scarcity and diversity, pp. 247–285. Sinauer Associates, Massachusetts.

MÜLLER, D., AND M. ZELLER. 2002. Land use dynamics in the central highlands of Vietnam: A spatial model combining village survey data with satellite imagery interpretation. Agric. Econ. 27: 333–354.

MUNROE, D. K., J. SOUTHWORTH, AND C. M. TUCKER. 2002. The dynamics of land cover change in western Honduras: Exploring spatial and temporal complexity. Agric. Econ. 27: 355–369.

MYNENI, R. B., W. YANG, R. R. NEMANI, A. R. HUETE, R. E. DICKINSON, Y. KNYAZIKHIN, K. DIDAN, R. FU, R. I. NEGRÓN JUÁREZ, S. S. SAATCHI, H. HASHIMOTO, K. ICHII, N. V. SHABANOV, B. TAN, P. RATANA, J. L. PRIVETTE, J. T. MORISETTE, E. F. VERMOTE, D. P. ROY, R. E. WOLFE, M. A. FRIEDL, S. W. RUNNING, P. VOTAVA, N. EL-SALEOUS, S. DEVADIGA, Y. SU and V. V. SALOMONSON. 2007. Large seasonal swings in leaf area of Amazon rainforests. Proc. Natl. Acad. Sci. USA 104: 4820–4823.

MYERS, N., R. A. MITERMEIER, C. G. MITERMEIER, G. A. B. DE FONSECA, AND J. KENT. 2000. Biodiversity hotspots for conservation priorities. Nature 403: 853–858.

PAINTER, R. L. E., R. B. WALLACE, AND H. GÓMEZ. 2006. Landscape conservation in the Greater Madidi Landscape in northwestern Bolivia: Planning for wildlife across different scales and jurisdictions. Case Study 2.2 *In* M. J. Groom, G. K. Meffe, C. Ronald Carroll and Contributors (Eds.). Principles of conservation biology, 3rd edition. Sinauer Associates, Massachusetts.

PICKETT, S., R. S. OSTFELD, M. SHACHAK, AND G. E. LIKENS. 1997. Ecological basis of conservation, heterogeneity, ecosystems, and biodiversity. Springer, New York, New York.

PINEDO-VASQUEZ, M., J. B. PASQUALLE, AND D. DEL CASTILLO TORRES. 2002. A tradition of change: The dynamic relationship between biodiversity and society in Sector Muyuy, Peru. Environ. Sci. Policy 5: 43–53.

SALATI, E., AND C. A. NOBRE. 1991. Possible climatic impacts of tropical deforestation. Climatic Change 19: 177–196.

SANCHEZ-AZOFEIFA, G. A., B. RIVARD, J. CALVO, AND I. MOORTHY. 2002. Remote sensing of forest change on the Osa Peninsula of Costa Rica. Mountain Res. Dev. 22: 352–358.

SANCHEZ-AZOFEIFA, G. A., G. C. DAILY, A. S. P. PFAFF, AND C. BUSCH. 2003. Integrity and isolation of Costa Rica's national parks and biological reserves: Examining the dynamics of land-cover change. Biol. Conserv. 109: 123–135.

SANDERSON, E. W., K. H. REDFORD, A. VEDDER, P. B. COPPOLILLO, AND S. E. WARD. 2002. A conceptual model for conservation planning based on landscape species requirements. Landscape and Urban Planning 58: 41–56.

SEDDON, N., J. TOBIAS, J. W. YOUNT, J. R. RAMANAMPAMONJY, S. BUTCHART, AND H. RANDRIANZAHANA. 2000. Conservation issues and priorities in the Mikea Forest of south-west Madagascar. Oryx 34: 287–304.

SILVA, R., D. ROBISON, S. MCKEAN, AND P. ÁLVAREZ. 2002. La historia de la ocupación del espacio y el uso de los recursos en el PNANMI Madidi y su zona de influencia. WCS-CARE and the European Community, La Paz, Bolivia.

SKOLE, D., AND C. TUCKER. 1993. Tropical deforestation and habitat fragmentation in the Amazon: Satellite data from 1979 to 1988. Science 260: 1905–1909.

STEELE, B. M., J. C. WINNE, AND R. L. REDMOND. 1998. Estimation and mapping of misclassification probabilities for thematic land cover maps. Remote Sens. Environ. 66: 192–202.

STEININGER, M. K., C. J. TUCKER, P. ERSTS, T. J. KILLEEN, Z. VILLEGAS, AND S. B. HECHT. 2001a. Clearance and fragmentation of tropical deciduous forest in the Tierras Bajas, Santa Cruz, Bolivia. Conserv. Biol. 15: 856–866.

STEININGER, M. K., C. J. TUCKER, J. R. G. TOWNSHEND, T. J. KILLEEN, A. DESCH, V. BELL, AND P. ERSTS. 2001b. Tropical deforestation in the Bolivian Amazon. Environ. Conserv. 28: 127–134.

THOMLINSON, J. R., P. V. BOLSTAD, AND W. B. COHEN. 1999. Coordinating methodologies for scaling landcover classifications from site-specific to global: Steps toward validating global map products. Remote Sens. Environ. 70: 16–28.

TOWNSHEND, J. R. G., V. BELL, A. C. DESCH, C. HAVLICEK, C. O. W. JUSTICE, W. L. LAWRENCE, D. SKOLE, W. W. CHOMENTOWSKI, B. I. MOORE, W. SALAS, AND C. J. TUCKER. 1995. The NASA Landsat Pathfinder Humid Tropical Deforestation Project. Land Satellite Information in the Next Decade, pp. IV76–IV87 American Society of Photogrammetry and Remote Sensing: Tyson's Corner, Virginia.

TUCKER, C. J., AND J. R. G. TOWNSHEND. 2000. Strategies for monitoring tropical deforestation using satellite data. Int. J. Remote Sensing 21: 1461–1471.

WALLACE, R. B., H. GÓMEZ, G. AYALA, AND F. ESPINOZA. 2003. Camera trapping capture frequencies for jaguar (*Panthera onca*) in the Tuichi valley, Bolivia. J. Neotrop. Mam. 10: 133–139.

WCS GREATER MADIDI LANDSCAPE CONSERVATION PROGRAM, 2005. Unpublished data. Análisis de cambios de uso y avance de la frontera agrícola.

WCS GREATER MADIDI LANDSCAPE CONSERVATION PROGRAM, 2007, Unpublished data. Collection database of the WCS Greater Madidi Landscape Conservation Program, Habitat types (1998–2004).

ZARIN, D. J., V. F. G. PEREIRA, H. RAFFLES, F. G. RABELO, M. PINEDO-VASQUEZ, AND R. G. CONGALTON. 2001. Landscape change in tidal floodplains near the mouth of the Amazon River. For. Ecol. Manage. 154: 383–393.

ZHANG, Q., D. DEVERS, A. DESCH, C. O. JUSTICE, AND J. TOWNSHEND. 2005. Mapping tropical deforestation in Central Africa. Environ. Monit. Assess. 101: 69–83.