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Assessing the impact of Celaque National Park on forest fragmentation in western Honduras

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Abstract

The effectiveness of parks as management regimes is much in debate. This study examines the effect of establishment of the Celaque National Park, Honduras, in 1987, on limiting deforestation through a comparison with the surrounding landscape using remote sensing, GIS and landscape pattern analysis. Pressure on the park region is found to relate spatially to the locations of towns and roads, with increasing deforestation in the landscape surrounding the park. In contrast, the park has been largely successful in maintaining forest cover. Although the extent of change within the park is not pronounced, the pattern of change is. Expansion of agriculture and coffee production have led to increasing pressure on the park boundaries, with as much as 25% of the landscape surrounding the park experiencing land cover change between 1996 and 2000. This has significant implications for the future of the park.

Keywords: National Parks; Remote sensing; GIS; Landscape Metrics; Agricultural intensification; Coffee

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Introduction

Over the past several decades, the environmental literature has increasingly talked of trends in tropical deforestation, and warned of serious consequences. One exceedingly popular "solution" employed to deal with this has been to create protected reserves that range from exclusive areas that attempt to prevent outside

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incursions, to relatively inclusive managed areas that involve local populations in management (Bates & Rudel, 2000; Brandon, Redford, & Sanderson, 1998). As the establishment of new protected areas continues to increase, there is increasing awareness of the social consequences of creating protected areas that exclude local inhabitants from traditional ways of life. The issue is particularly acute in tropical regions where people live in and depend upon high biodiversity regions for their livelihoods. Increasingly, the effectiveness of protected areas at halting degradation is being called into question. "Paper parks" that exist only in name fail to protect resources and "sustainable use" has been proposed as a more effective alternative, based on the argument that resources are better conserved when people can use and therefore value them increasingly (Bates & Rudel, 2000; Brandon et al., 1998).

Research on the scientific basis for reserve selection and design has developed rapidly (e.g. Bishop, Phillips & Warren, 1995; Schwartz, 1999). There is an equal need for studies that have addressed the effectiveness of protected area establishment using empirical data. The few large-*N* comparative studies that exist (notably Bates & Rudel, 2000; Bruner, Gullison, Rice, Gustavo, & da Fonseca, 2001) suggest that the creation of protected areas has had mixed outcomes. Although some broad trends can be discerned, the reasons for the success or failure of a particular park are often locality-specific. Although there is a clear need for such valuable large '*n*' analyses to be supplemented by careful individual case studies, there has been limited research that directly examines and contrasts land cover before and after the establishment of a park (Liu, Linderman, Ouyang, An, Yang, & Zhang, 2002).

Remote sensing provides a particularly effective tool for such an analysis. The most frequently used technique for the mapping of changes in tropical forests is by the analysis of satellite data. Such data from several time points allows the creation of land cover maps over greater spatial extents and more frequent time steps than is possible with expensive and detailed field studies (Jensen, 2000; Nagendra, 2001). While direct loss of forest area is a primary concern, forest fragmentation issues also assume vital significance in the context of maintaining "natural" variability in the mosaic of patches within a landscape (Riitters, Wickham, O'Neill, Jones, & Smith, 2000; Sanchez-Azofeifa, Daily, Pfaff, & Busch, 2003). Since these classifications are spatially explicit, they not only provide information on percent changes in forest cover, but also allow for evaluation of changes in landscape spatial pattern and fragmentation over time, factors with crucial implications for biodiversity (Forman, 1995; Rivard, Poitevin, Plasse, Carleton, & Currie, 2000; Parks & Harcourt, 2002).

In the past few decades, the increase in the number of protected areas in Latin America has been particularly dramatic (Bates & Rudel, 2000). Indeed, the tropical rain forests located within protected areas in Latin America have increased from less than 200,000 km^2 in 1965, to over 1,100,000 km^2 in 1990 (Harcourt & Sayer, 1996). We choose one of these protected areas, the Celaque National Park in western Honduras, to conduct an empirical case study on the effectiveness of park establishment in halting deforestation. There are several reasons for selection of this area. First, Honduras is one of the poorest nations in Latin America (World

Bank, 2001), and the area where the Celaque National Park has been established faces a high demand for land: it thus typifies the conflict between land for development and conservation in developing nations throughout the world. Second, the park contains the highest mountain peak in Honduras and its cloud forests are rich in biodiversity and endemic species (Fonseca, Moreno, & Padgett, 1999; Humphrey, 2000). However, this region has been hitherto little studied due in large part to its inaccessibility. Third, due to previous research in the region surrounding Celaque (Southworth & Tucker, 2001; Southworth, Nagendra, & Tucker, 2002), we possess valuable information on land use changes in this region that helps inform our interpretations of land cover change. Finally, this park was established relatively recently. We are able to use satellite imagery of constant spatial, spectral and radiometric resolution (Landsat TM imagery) to examine whether landscape fragmentation and land cover change differs within and outside park boundaries since the park's creation, and thus gain fundamental information to evaluate park effectiveness. Such analysis is difficult to do, as it is often impossible to obtain remote sensing data with constant characteristics over extended time scales due to changes in the sensors (Liu et al., 2002).

This study addresses the effectiveness of park boundaries in deterring deforestation within the Celaque National Park in western Honduras, a protected area of approximately 266 km^2. Most forest reserves or protected areas are ecosystem remnants of a limited size. Few, if any, represent intact ecosystems and hence, it has become increasingly important to locate each protected area as a functional component of a larger landscape (Mladenoff, White, Pastor, & Crow, 1993; Rivard et al., 2000; Parks & Harcourt, 2002). This research therefore compares the condition of the park to the larger landscape within which the park is embedded, in order to investigate whether the extent and pattern of change differ between the protected and unprotected landscapes. In this remote and inaccessible region, remote sensing may be the only feasible method to monitor forest change over time. Our analysis utilizes information from a larger study area in which we have extensive field data, socioeconomic information (Tucker, 1999a, b), and satellite derived land cover classifications across four dates (Southworth & Tucker, 2001; Southworth et al., 2002; Southworth, Munroe, & Nagendra, 2004). This allows us to associate changes in land cover pattern with information on the causal processes of land use change in this complex, dynamic landscape.

Materials and methods

Site description

Western Honduras is characterized by mountainous topography with slopes averaging over 30°. Soils tend to be thin and rocky; 97% of the region is considered unsuitable for intensive agriculture (Chavez Borjas, 1992) (Fig. 1). Celaque is the tallest mountain in Honduras with its highest peak at 2849 m above sea level (Pineda Portillo, 1997). In 1987, the country began a program of rapid expansion of the protected area system, with National Law 87–87 declaring all areas above

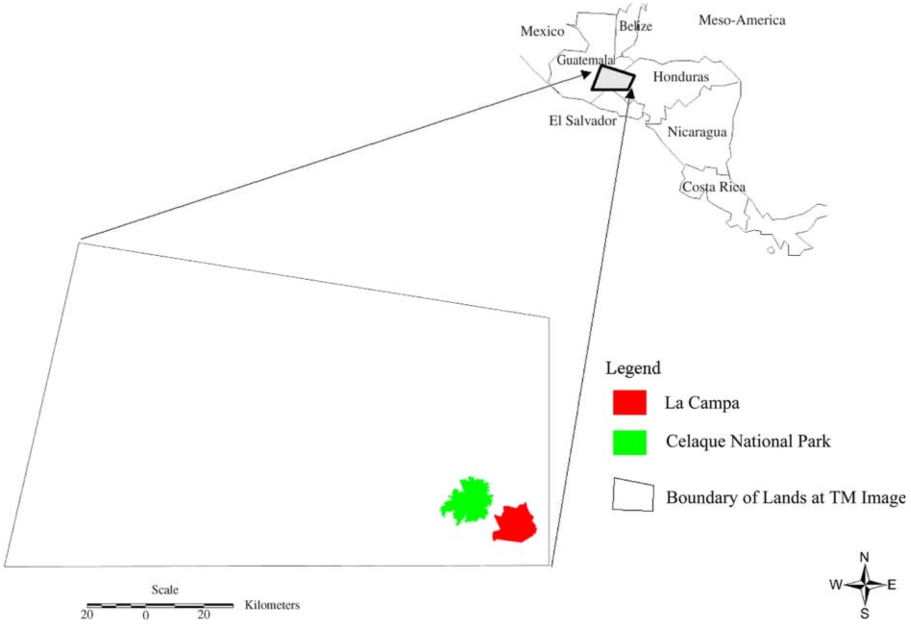

Fig. 1. Study area map showing specific areas of interest: Celaque National Park, and La Campa, and the location of the image area within Meso-America.

1800 m, including Celaque National Park, to be protected (Pfeffer, Schelhas & Day, 2001). A core zone was established above 1800 m elevation with an area of approximately 165 km^2, in which agriculture and industry was prohibited (Fig. 2). Subsequently, in response to the recognition that there were small communities that still lived in the vicinity of the park, an outer park boundary was created (Fig. 2). There are currently approximately 165 km^2 within the core zone, with eight communities still living here. The park boundary includes approximately 266 km^2, and an additional 26 communities (estimated at 2800 people) reside within and just outside these park limits (Aguilar, 2003; Fonseca et al., 1999).

The park ranges in elevation from 1000 to 2849 m above sea level. Topography is rugged, with over two-thirds of the land having slopes greater than 60%, and soils generally being sandy and shallow (Archaga, 1998a). In the language of the local Lenca inhabitants, Celaque means “box of water”: this can be attributed to the fact that 11 major rivers and a number of streams originate from Celaque (Humphrey, 2000). The floristic zones vary with elevation. Below 1800 m, the park consists mainly of *Pinus–Quercus* (pine–oak) forests. At higher elevations, the pine–oak forest gives way to a transition mixed broad-leaf/pine montane forest. Above 2200 m, true cloud forest begins with many broadleaf species (Archaga, 1998a).

The park area is shared among the departments of Lempira, Ocotepeque and Copan. The main entrance to the park is located approximately 9 km southwest of

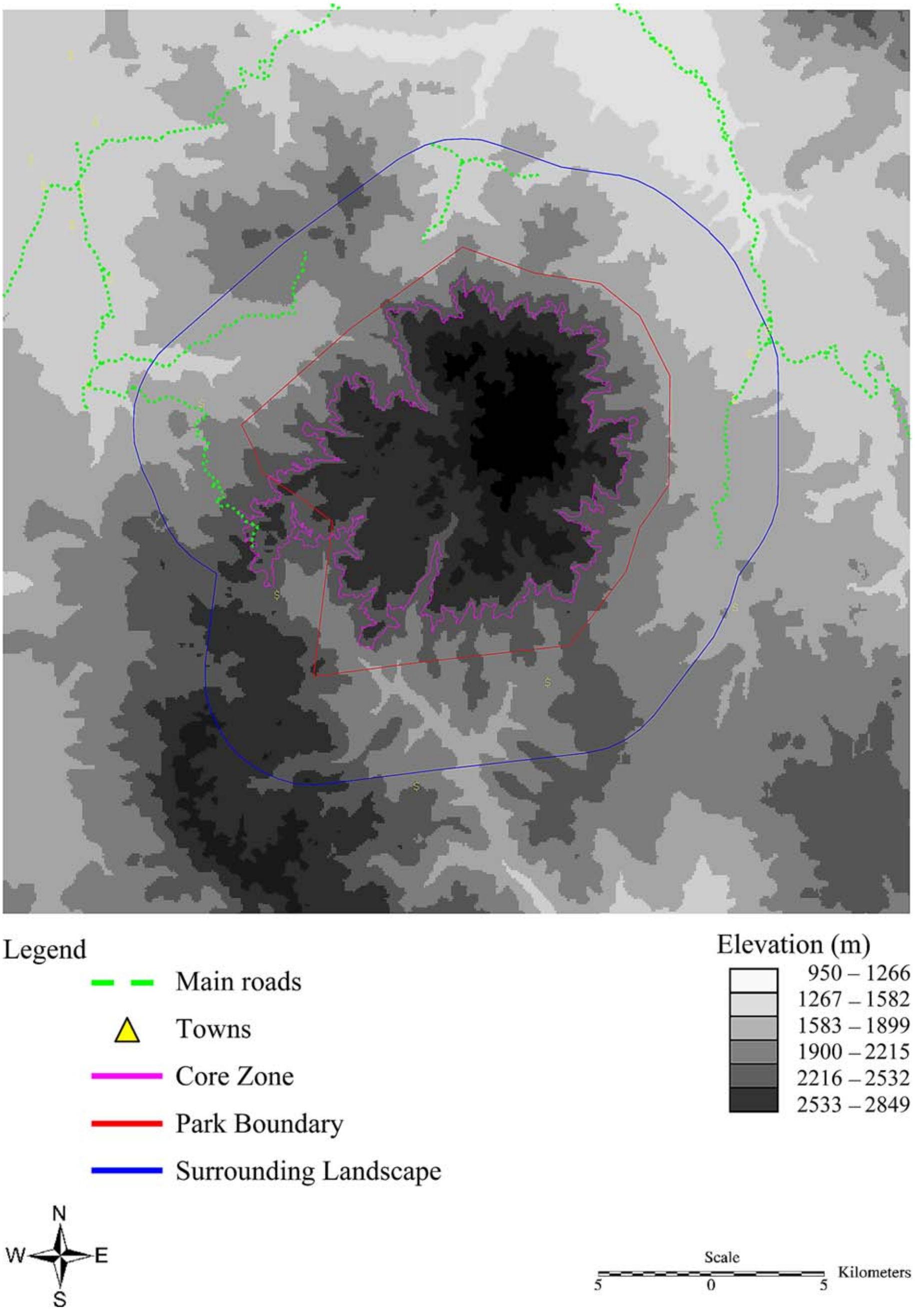

Fig. 2. The Celaque National Park core zone (pink), park boundary (red), and surrounding landscape (blue) overlaid on elevation (grey shade), main roads (in green) and towns (in yellow) (For interpretation of the references to colur in this figure legend, the reader is referred to the web version of this article.).

Gracias, the capital of Lempira (Humphrey, 2000). Several non-governmental organizations (NGOs) work with the communities bordering the park (Fonseca et al., 1999). According to 1998 estimates, a population of 69,680 resides in the *municipios* surrounding the park, with an estimated growth rate of 3% per annum (Fonseca et al., 1999).

Approximately 80% of the population residing within the park is of indigenous Lenca Indian origin (Aguilar, 2003; Fonseca et al., 1999). Shared ethnic identity appears to be an underlying factor that has contributed to social solidarity, the preservation of common property land, and a strong local government. Most households depend primarily upon subsistence production of maize, beans, wheat and sugarcane for their livelihood. More than 6% of the area within the park is used for subsistence agriculture (Archaga, 1998b). Deforestation continues, especially on the east, west and north regions where the expansion of coffee plantations during the past decade represents a notable factor in land cover change and increasing social heterogeneity (Nagendra, Southworth, & Tucker, 2003). Illegal logging, agricultural clearing and coffee plantations and other farms are frequent in the park and buffer zone (Japan International Cooperation Agency, 2002.

According to a study by Fonseca et al. (1999), 96% of the population sampled did know of the park's existence. Most of these people learned about the park through the NGOs working in the park, through friends, and through the radio, rather than through government agencies. But while people did know about the park, less than 30% knew the location of the boundary and core. Even the park rangers were not well informed about the protected areas of Honduras, or the flora and fauna within this region.

As with much of the MesoAmerican region (Aguilar-Stoen & Dhillion, 2003) there have been few prior studies on the Celaque region that attempt to quantify the biodiversity, or the extent and pattern of land cover change in and surrounding the park (Aguilar, 2003 being the only published study we know of). Part of the difficulty of studying Celaque relates to its inaccessibility; remote sensing is thus a most effective means for studying this region. Previous research on forest cover change in the larger landscape that surrounds Celaque National Park found a trend towards reforestation between 1987 and 1996 (Southworth & Tucker, 2001; Southworth et al., 2002). This apparent reversal in the dominant trend of deforestation in Honduras is puzzling in the context of the general trend towards deforestation with increase in population pressure (Mather & Needle, 2000). Given this regional context, it is especially interesting to look at whether the park contrasts from the surrounding area in land cover change, which can provide an indication of park effectiveness.

Image analysis pre-processing

Landsat TM images were obtained for March 1987, 1991, 1996, and 2000. This corresponds to the end of the dry season when fallow agricultural lands can be easily distinguished from forests. Images were geometrically registered to 1:50,000 survey topographic maps of the region, followed by radiometric calibration and atmospheric correction. Without such calibration, change detection analysis may evaluate differences at the sensor level rather than changes at Earth's surface (Jensen, 2000).

Image classification and change detection

Training sample data were used to determine the land cover classes on the ground and then train a supervised classification of the satellite image. In order to simplify the land cover change analysis (Kleinn, Corrales & Morales, 2002) we used two cover classes at each date. Classes for agriculture, young fallows (approximately 1–3 years), cleared areas, bare soil, water, and urban areas were aggregated in this analysis to create a non-forest class. Land cover maps of forest and non-forest cover for 1987, 1991, 1996, and 2000 were derived by independent supervised classification of the Landsat images, using a Gaussian maximum likelihood classifier (Southworth et al., 2002; Southworth & Tucker, 2001).

This data can be used to create change trajectories, i.e., sequences of successive changes in land cover types (Petit, Scudder, & Lambin, 2001). This technique is used to determine the change between two or more time periods of a particular region or for a particular land cover, and provides quantitative information on spatial and temporal distribution (Macleod & Congalton, 1998; Mertens & Lambin, 2000). Nevertheless, few researchers have explicitly utilized change trajectories in analyses of land cover change and landscape fragmentation (Helmer, 2000; Nagendra et al., 2003; Petit et al., 2001). Using a GIS, the individual land cover images for 1987, 1991, 1996, and 2000 were analyzed to identify change trajectories. The greater the number of land cover types or classes, and dates, the greater the number of change trajectories. Thus,

$$m_{\mathrm{t}} = m_{\mathrm{c}}^{t}$$

where m_{t} is the number of trajectories, m_{c} is the number of land cover classes and superscript t is the number of images in the temporal series (Petit et al., 2001). With four dates and two classes in each date, there are a potential total of 16 change classes, and interpretation can get confusing. To facilitate ease of interpretation, and to separately analyze each time period, we use a sequence of two dates at a time, and therefore four change classes per change period. To illustrate, a change analysis between 1996 and 2000 will create four land cover change classes: areas that are forested in both dates (F-F), areas cleared in both dates (NF-NF), areas that have experienced deforestation during this time period (F-NF), and areas that have undergone reforestation (NF-F). To create change trajectories between pairs of dates (1987–1991, 1991–1996, 1996–2000), the forest/non-forest classifications were overlaid using ARC/INFO™ GRID software, resulting in four change classes for each pair of dates.

In this region, reforestation is primarily associated with regrowth in abandoned agricultural fields and logged patches, and with active planting of shade trees for shade grown coffee. Regrowth of vegetation is fairly rapid and field verification indicates that we are indeed able to identify regrowth in the satellite image within a time period of 4 years (Nagendra et al., 2003). Single-date classification accuracies of above 85% were obtained, along with kappa statistics above 0.75. This is fairly high considering the mountainous nature of our landscape, which can make classification difficult, and compares very favorably with land cover classification accuracies obtainable for Central America (Sader, 1995). Further, for the land cover change trajectories, we have also demonstrated an overall accuracy of 92.6%. This high degree of accuracy is likely to be due to the fact that it is very easy to differentiate forested area from bare soil during the dry season in which all images were collected.

GIS analysis

The digitized park boundary and core zone boundaries (>1800 m) were obtained from the Friends of Celaque (FOC) research group. Note that the park boundary includes all areas within the park except for an area on the west side where the boundary does not include a small region of 1800 m elevation plus. This is due to the location of a settlement that predates park establishment and was too large to be considered for relocation. Hence, the boundary simply excludes this area, but the core zone, which is based solely on elevation (>1800 m), does include this region.

In addition, a GIS buffer region was created that extended 5 km beyond the outer park boundary polygon. This allowed for a comparison of land cover change within the boundary and core, to change in the surrounding landscape. This is a necessary step that allows us to ground our results within the broader context of land cover change within the region, and without it, the impact of park establishment is difficult to evaluate. For instance, heavy clearing of forest around but not within the park would indicate that there is a high level of human pressure on forests in the region. In this context, if the park is maintaining forest cover, this implies that the park boundaries are effective, i.e., the change only occurs outside the park. However, if deforestation occurs to an equal extent within and outside the park, then park creation would be deemed ineffective. The choice of the buffer distance (5 km) is admittedly somewhat arbitrary, but was made keeping in account the decreasing elevation with increasing distance from the park, such that the landscape would include higher elevations more comparable to the park. A 1:50,000 DEM and GIS coverages of roads, towns, elevation, and slope were also overlaid on the classified images to facilitate our interpretation of the causes of land cover change.

Three subsets of the classified images were created using the coverages of the park core zone (all areas above 1800 m elevation, hereafter referred to as "core"), the official park boundary (this includes the park boundary, and most of the core region, as described above and is hereafter referred to as "boundary"), and the surrounding 5 km landscape buffer (hereafter referred to as "surrounding landscape"). This allows for the comparison of the amount of land cover change from 1987 to 1991, 1991 to 1996, and 1996 to 2000 for each separate subset.

Landscape metrics

Landscape metrics were calculated using the software Fragstats 2.0 (McGarigal & Marks, 1994) for change coverages in each subset: core, boundary, and surrounding area. Fragstats provides a very comprehensive set of spatial statistics and descriptive metrics of pattern at the patch, class, and landscape levels (Haines-Young & Chopping, 1996). Landscape pattern indices provide useful quantification to better analyze study site variability and changes over time (Imbernon & Branthomme, 2001). Several of these metrics are partially or wholly redundant. For instance, there are several measures of patch shape. At the patch level, indices of size (area in hectares), shape (computing the complexity of patch shape, compared with a square patch of identical area, taking values of 1 when most compact and increasing without limit as the patch becomes more irregular), and inter-patch distance (distance from a patch to the nearest neighboring patch of the same cover type, in meters) were considered to quantify distinct aspects of patch structure (Haines-Young & Chopping, 1996; McGarigal & Marks, 1994). One-tailed Mann–Whitney *U*-tests (Sokal & Rohlf, 1981) were used to assess whether patch size differed significantly ($p = 0.05$), across dates, and between categories of land cover change. Tests of significance for differences in patch shape and nearest neighbor distance were also similarly carried out.

At the class level, our interest is in comparing descriptive metrics of land cover pattern between forest and non-forest classes, and across categories of land cover change. These indices can be grouped into categories of area, shape, nearest neighbor, core, diversity, and contagion/interspersion (Haines-Young & Chopping, 1996). To simplify interpretation, the following indices were considered, as believed to quantify different aspects of structure (Haines-Young & Chopping, 1996):

(a) Largest patch index (LPI): the area of the largest patch in each class (in hectares).
(b) Number of patches (NP): the total number of patches in this category.
(c) Mean patch size (MPS): average patch size or area for the class (in hectares).
(d) Nearest neighbor (NN): edge–edge distance between a patch and the nearest neighboring patch of the same class (in meters).
(e) Edge density (ED): sum of length of all edge segments, divided by total area for each class.
(f) Mean shape index (MSI): measures the average complexity for a category, of patch shape, compared to a square patch of identical area. For a single patch, the shape index is 1 when square, and increases without limit as the patch becomes more irregular.
(g) Interspersion-juxtaposition index (IJI): measures the degree of interspersion of patches of this class, with all other categories. This index takes values from 0 to 100, decreasing as the distribution of patch adjacencies among types becomes increasingly uneven.

Complete descriptions of these metrics and equations for their calculation are provided in McGarigal and Marks (1994). The indices of LPI, NP and MPS correspond to area metrics. Together with NN and ED, these provide indications of the degree of fragmentation for different land cover types and land cover change trajectories. MSI and IJI provide metrics of shape and contagion/interspersion. This analysis does not include measures of core (we do not know of an ecological basis for defining core distance in this landscape) or diversity (as we use a constant number of classes across time, diversity indices do not vary).

Results

Extent of land cover change

Individual forest/non-forest image classifications were created for 1987, 1991, 1996, and 2000. Overall classification accuracies for the single-date classifications were above 85%, with kappa statistics greater than 0.75, for all dates, and for the change trajectories were 92.6% (see Nagendra et al., 2003 for details regarding accuracy assessment). This is an acceptable level of accuracy for the purpose of land cover change analysis (Thomlinson, Bolstad, & Cohen, 1999). Table 1 describes the extent of forested and non-forested land cover in the core, boundary, and the surrounding landscape, during 1987, 1991, 1996, and 2000. As a percentage, most of the area in the core, and to a lesser extent within the boundary, is forested for all dates. In contrast, nearly half of the area in the surrounding landscape is non-forested. In the core and boundary, the area under forest cover appears to have remained fairly static across all four dates of analysis. Indeed, the core loses less than 1 km^2 of forest cover between 1987 and 2000, while the boundary loses about 13 km^2. In contrast, the area under forest cover in the surrounding landscape remains fairly stable between 1987 and 1996, it then decreases sharply between 1996 and 2000.

The data in Table 1 describe the aggregate forest and non-forest cover in each year, and may create a somewhat more static impression of forest cover than is the case. A constant amount of overall forest cover may mask significant ongoing

Table 1
Extent of forested and non-forested area (km^2) of Celaque National Park, for the area within the core, the boundary, and the surrounding landscape, assessed using Landsat TM satellite imagery from 1987, 1991, 1996, and 2000

Year	Class name	Core zone (km^2)	Park boundary (km^2)	Surrounding landscape (km^2)
1987	Forest	144.75	217.07	198.95
	Non-forest	20.79	46.68	197.79
1991	Forest	149.64	224.12	188.74
	Non-forest	15.91	41.60	208.01
1996	Forest	144.55	218.44	199.42
	Non-forest	20.99	47.31	197.33
2000	Forest	143.82	203.99	148.98
	Non-forest	21.71	61.73	247.76

deforestation, if balanced by an equal amount of reforestation in other parts of the landscape. It is therefore necessary to conduct an analysis of land cover change trajectories to examine change over time in further detail. For the area within the core, boundary, and surrounding landscape, Table 2 describes the area occupied by the four change categories of stable forest (F-F), reforestation (NF-F), deforestation (F-NF), and stable agriculture (NF-NF), across three time periods (1987–1991, 1991–1996, and 1996–2000). The area within the core and boundary appear relatively stable, with less than 10–12% of the total area undergoing change at any point in time. In contrast, the surrounding landscape is much less stable, with as much as 25% of the area experiencing change (reforestation or deforestation) between 1996 and 2000.

Between 1987 and 1991, and 1991 and 1996, for core, boundary, and surrounding landscape, the area in stable forest and stable agriculture stays approximately the same (± 1 km^2). In the most recent time period of 1996–2000, however, there is a sharp decrease in the area under stable forest cover, and a corresponding increase in the area under stable agriculture in all three regions. Trends of deforestation/reforestation vary with location. Within the core, more than 10 km^2 is reforested between 1987 and 1991: this declines to less than 5 km^2 in 1991–1996, and then rises slightly to about 8 km^2 between 1996 and 2000. Conversely, there is minimum deforestation between 1987 and 1991, followed by high deforestation between 1991 and 1996, with a subsequent decrease between 1996 and 2000. Within the boundary there is a consistent trend of decreasing reforestation and increasing deforestation through time. In the surrounding landscape, reforestation rates initially rise from 1987–1991 to 1991–1996, and then drop again. Deforestation rates follow an opposite trend, declining from 1987–1991 to 1991–1996, but then increasing again between 1996 and 2000. The amount of area deforested in the surrounding landscape between 1996 and 2000 is 72.93 km^2, twice the extent of area deforested between 1991 and 1996 for this region (Table 2).

Spatial patterns of land cover change

Visual observations of Figs. 2 and 3 demonstrate that most of the stable agriculture and deforestation occurs along the southern and eastern edges of the park. This relates to the location of towns (Fig. 2) and to a lesser extent to roads. The areas of highest elevations are the areas predominantly in forest. However, small patches of clearing and reforestation do occur within these regions (Fig. 3).

A one-tailed Mann–Whitney U-test was used to analyze differences in statistics of patch pattern across different time points. The area of the stable forest and stable agriculture patches remains relatively constant through time. However, patches of reforestation decrease significantly in area over time ($p < 0.05$), and are smallest in 1996–2000 for core, boundary and surrounding landscape; conversely, patches of deforestation increase in size over time, and are largest during 1996–2000 (Table 3). Statistics of patch shape more or less parallel to those of patch area. There is no significant change in NN distance between patches of stable forest and stable agriculture over time. However, the distance between patches of

Table 2
Extent of area occupied by land cover change categories (km^2) of Celaque National Park, for the area within the core, the boundary, and the surrounding landscape, assessed using Landsat TM satellite imagery, during three time periods: 1987–1991, 1991–1996, and 1996–2000

Year	Change category	Change category name	Core zone (km^2)	Park boundary (km^2)	Surrounding landscape (km^2)
1987–1991	F-F	Stable forest	139.44	206.71	154.45
	NF-F	Reforestation	10.20	17.41	34.29
	F-NF	Deforestation	5.32	10.35	44.51
	NF-NF	Stable agriculture	10.58	31.25	163.50
1991–1996	F-F	Stable forest	139.58	207.60	153.09
	NF-F	Reforestation	4.98	10.83	46.32
	F-NF	Deforestation	10.06	16.52	35.65
	NF-NF	Stable agriculture	10.92	30.77	161.68
1996–2000	F-F	Stable forest	135.89	191.61	126.49
	NF-F	Reforestation	7.92	10.38	22.49
	F-NF	Deforestation	8.61	24.77	72.93
	NF-NF	Stable agriculture	13.11	39.96	174.84

reforestation increases over time in all three zones ($p < 0.05$), with patches being farthest apart in 1996–2000; conversely, the NN distance between deforestation is the least in 1996–2000 (Table 3).

Differences in patch spatial pattern were also observed between land cover change categories. In all three time periods, irrespective of location, patches of stable agriculture are significantly larger than patches of stable forest, deforestation and reforestation (with the exception of the area within the surrounding landscape in 1996–2000). During 1987–1991, and 1991–1996, patches of deforestation are smaller in area compared to patches of forest and of reforestation. Subsequently, there is a switch in spatial pattern. In 1996–2000, patches of deforestation increase and are for the first time significantly larger than patches of stable forest and reforestation ($p < 0.05$). Patch shape statistics are parallel to those of patch size, and no consistent differences between classes could be discerned in terms of NN distance (Table 3).

The core and boundary are dominated by one large stable forest patch, which decreases in area over time, mostly through loss of area at the edge (Table 1). Patches of stable forest within the core and boundary have a larger MPS, lower mean NN distance, greater edge density and a higher IJI compared to patches in the surrounding landscape. Patches of stable agriculture inside the core and boundary have a lower MPS, lower mean NN distance and greater edge density compared to the surrounding landscape; interspersion-juxtaposition values do not vary appreciably with location. As with the patch-level statistics, the class-level statistics demonstrate that patches of reforestation and deforestation are much smaller in size and larger in number compared to stable forest and stable agriculture. Over time, the mean NN distance between patches of reforestation increases, and for deforestation decreases, for all locations.

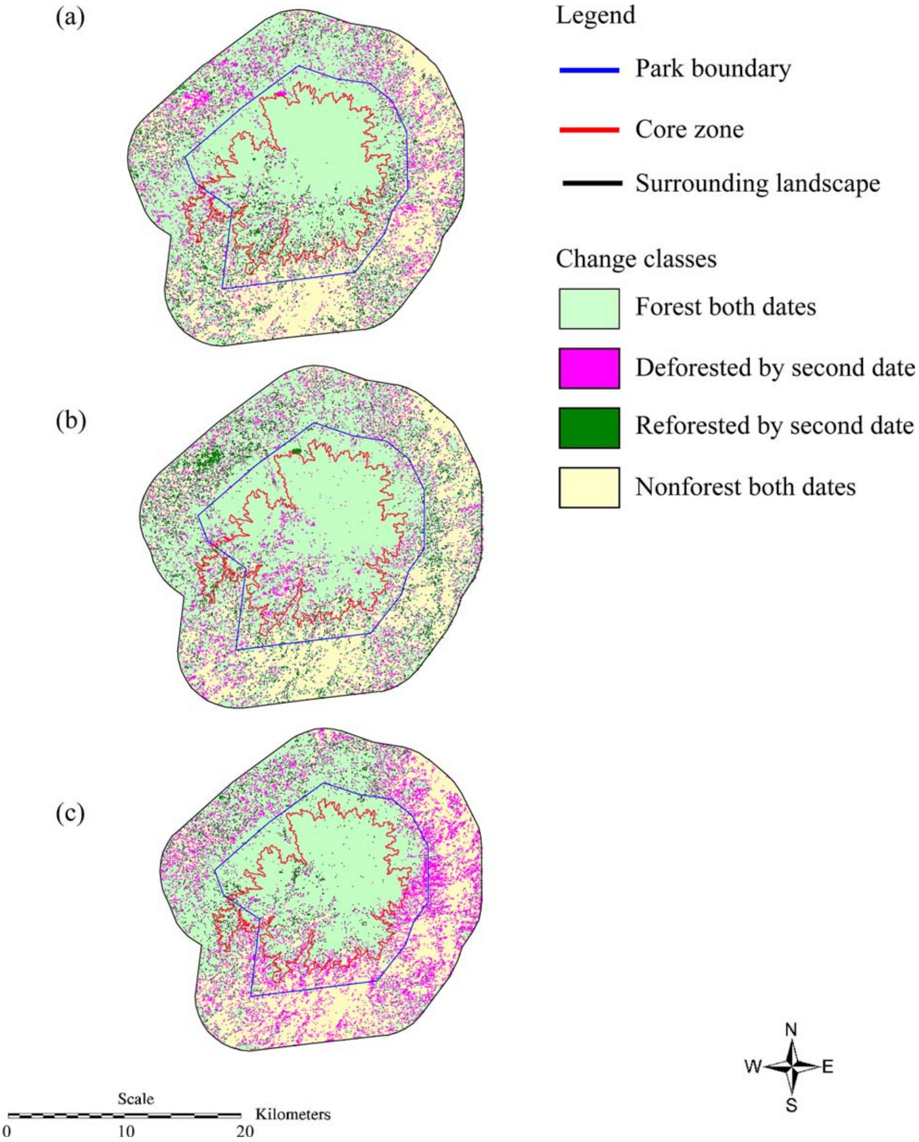

Fig. 3. Land cover change classes for (a) 1987–1991, (b) 1991–1996, and (c) 1996–2000 for the Celaque National Park core zone, park boundary, and the surrounding landscape.

Discussion and conclusions

The Celaque National Park is a highly biodiverse area, with rare and endemic species present in its higher altitude forests (Fonseca et al., 1999). Owing to the high topography and relative inaccessibility, little is known about land cover

Table 3
Metrics of landscape pattern, for land cover change classes in the Celaque National Core, boundary, and surrounding landscape, during three time periods: 1987–1991, 1991–1996, and 1996–2000

Year	Location	Change category	LPI	NP	MPS (ha)	NN (m)	ED	MSI	IJI
1987–1991	Core	F-F	82.4	266	52.4	45.9	49.5	1.35	85.4
		NF-NF	0.8	772	1.4	76.8	26.6	1.36	96.7
		NF-F	0.2	1980	0.5	57.3	43.9	1.30	73.3
		F-NF	0.1	1516	0.4	68.2	24.8	1.21	84.6
	Boundary	F-F	75.3	611	33.8	46.2	55.9	1.33	88.5
		NF-NF	3.1	1185	2.6	75.6	35.7	1.35	97.1
		NF-F	0.1	3884	0.5	51.9	50.3	1.29	78.0
		F-NF	0.1	3293	0.3	60.0	32.2	1.21	87.4
	Surrounding Landscape	F-F	17.7	2705	5.7	54.4	89.6	1.39	95.2
		NF-NF	15.6	2829	5.8	55.8	83.8	1.32	95.3
		NF-F	0.1	9629	0.4	50.1	70.8	1.24	86.7
		F-NF	0.4	9859	0.5	47.0	83.8	1.27	84.8
1991–1996	Core	F-F	82.4	237	58.9	45.0	51.5	1.37	84.9
		NF-NF	0.8	792	1.4	78.9	27.6	1.35	98.0
		NF-F	0.1	1694	0.3	63.5	25.9	1.21	86.7
		F-NF	0.2	2538	0.4	64.7	45.3	1.22	72.3
	Boundary	F-F	75.8	570	36.4	44.2	59.4	1.34	89.1
		NF-NF	3.7	1260	2.4	76.5	37.0	1.34	98.5
		NF-F	0.1	3482	0.3	57.7	35.4	1.23	87.1
		F-NF	0.1	4753	0.4	57.8	50.2	1.22	75.8
	Surrounding Landscape	F-F	18.8	2832	5.4	52.2	95.7	1.37	95.6
		NF-NF	15.8	3241	5.0	55.3	88.1	1.32	94.6
		NF-F	0.6	10240	0.5	46.2	88.9	1.28	84.5
		F-NF	0.1	10292	0.4	49.5	73.4	1.23	85.7
1996–2000	Core	F-F	80.2	295	46.1	43.6	49.9	1.34	86.4
		NF-NF	0.9	744	1.8	69.9	29.2	1.36	92.5
		NF-F	0.2	2325	0.3	71.2	37.9	1.2	72.9
		F-NF	0.1	1582	0.5	63.1	36.5	1.33	83.9
	Boundary	F-F	70.8	672	28.8	54.2	52.2	1.31	87.5
		NF-NF	4.5	1246	3.0	62.7	41.5	1.35	75.8
		NF-F	0.1	3481	0.3	73.2	32.7	1.18	74.4
		F-NF	1.2	2614	1.0	56.9	1.4	1.38	78.9
	Buffer Surrounding Landscape	F-F	24.0	2494	5.1	63.0	74.4	1.33	90.0
		NF-NF	18.1	3003	5.8	52.5	88.9	1.33	69.4
		NF-F	0.1	6995	0.3	60.9	46.7	1.20	85.3
		F-NF	1.3	7580	1.0	45.5	108.2	1.38	75.8

change in this region (Aguilar, 2003; Southworth & Tucker, 2001). Given the recent creation of the park (1987) our change analysis addresses whether the park has experienced distinct patterns of change as compared to the surrounding landscape since the park's inception. There is much current controversy on parks as management regimes (Bruner et al., 2001). Although the literature on the effects of protected area establishment on local livelihoods is well documented, there is an equal need for case studies that examine whether parks experience different rates and patterns of land cover change as opposed to non-protected areas. Such studies are integral to determining the effect of park establishment on ecological degradation (Hayes, Sader, & Schwartz, 2002; Sader, Hayes, Hepinstall, Coan, & Soza, 2001; Sanchez-Azofeifa, Rivard, Calvo, & Moorthy, 2002; Sanchez-Azofeifa, Daily, Pfaff, & Busch, 2003).

Our results indicate there is increasing pressure being applied to this park region, with increasing deforestation taking place in the surrounding landscape outside the park. The time period of maximum change is between 1996 and 2000, with as much as 25% of the surrounding landscape outside the park experiencing alteration in land cover. Nevertheless, the area within the boundary, and particularly inside the core, appears to be maintaining forest cover. Pressure on the park region is spatially variable and is related to the locations of towns, and to a lesser extent to roads. Hence, much of the pressure is on the south, south-west and south-east sides of the park.

The core relates to areas above 1800 m elevation, with very steep slopes (Archaga, 1998a). This topographic inaccessibility probably explains a large part of why the park has thus far remained relatively well forested. Given the increasing conversion to mountain grown coffee, and the trend towards agricultural extensification (Uebelhör, 1998), the pressure within the boundary is now increasing, as also indicated by the change in spatial pattern within the park. In order to ensure park survival, there is a clear need to involve local inhabitants with conservation efforts. While local NGOs are working with local populations, their efforts need to be supplemented by involvement of the Honduran government agencies (Fonseca et al., 1999).

Although the area inside the boundary and core maintains a nearly constant extent of forest cover, they experience a significant change in landscape pattern during the time period 1996–2000. Patches of reforestation are smallest and furthest apart in this time period than they have been in previous years, while patches of deforestation are largest and closest together during this period. This is most likely related to an increase in the area cut down for planting coffee. As in other parts of Central America (Daily, Ehrlich, & Sanchez-Azofeifa, 2001), coffee in these areas was grown initially as small clearings at higher elevations and on steeper slopes, which expand in area during subsequent years. In a previous study of the larger landscape including Celaque National Park (Nagendra et al., 2003), we had indicated that we expected to see a expansion in patch size of deforested areas after 1996: the findings reported here support this conclusion. In addition, the establishment of the park does not mean that communities residing within the boundary have ceased to practice agriculture; 6% of the park area is dedicated to small-plot subsistence agriculture (Rivas, 1994).

There has been considerable interest in understanding the impact of protected area establishment on land cover change in Central America. In a study of the Mayan Biosphere Reserve in Guatemala, Sader et al. (2001) found that the highest rates of forest clearing occurred in the buffer zone, and were in closer proximity to roads and rivers. A further study in the same region (Hayes et al., 2002) corroborated these findings, indicating the continued increase in rates of forest clearing from 1974 to 1997, with the highest rates of clearing taking place in the most recent time period. In Nicaragua, a study of land cover change in the BOSAWAS Natural Resource Reserve Region also indicated increased deforestation over time due to clearing for agriculture. This is similar to the conclusions we derive from our analysis of Celaque National Park.

Schelhas (1991) found extensive deforestation and conversion to pasture in the lands adjacent to Braulio Carrillo National Park, Costa Rica. This was attributed to waves and cycles of colonization that drive people to settle forested lands. While we find similar pressure on the land adjacent to Celaque National Park, colonization due to external migration is not significant in this region. As in the lowlands of Costa Rica (Schelhas, 1996; Thacher, Lee, & Schelhas, 1997), we also find the pressure for conversion to coffee to be a significant factor driving deforestation in our study area in the mountains of Honduras. In extensive studies of land cover change in and around protected areas in Costa Rica, Sanchez-Azofeifa et al. (1999, 2002, 2003), however, conclude that although rates of deforestation in Costa Rica continue to be alarmingly high, the establishment of protected areas has dramatically curtailed the rate of deforestation and forest fragmentation inside reserves. Nevertheless, they also find that pressure on these parks appears to be increasing as highlighted by rapid increases in deforestation as one ventures a little outside the reserve.

Such research highlights the importance of addressing issues of spatial pattern in addition to basic information on amounts of change, and the utility of remote sensing analyses to provide objective information on land cover and spatial patterns of change. In conjunction with socio-economic information on the driving processes of land use, satellite analyses of land cover change can prove invaluable for environmental monitoring of change (e.g. Hayes et al., 2002; Sader et al., 2001; Southworth et al., 2002).

One multiple-case study that addresses this issue (Bruner et al., 2001) concludes that tropical parks seem surprisingly effective at stopping land clearing. However, given the broad scale of this analysis, the parks selected varied significantly in size, management strategy, and severity of threat. There is a clear need for such research to be supplemented by detailed case studies; however, as a recent paper on China (Liu et al., 2002) points out, "there is little research comparing ecological degradation before and after the protected areas were established". Liu et al. (2002) analyze forest fragmentation in the Woolong Nature Reserve for giant pandas, in China, and conclude that rates of habitat loss and fragmentation inside the reserve have actually increased since the reserve was established, and now compare to fragmentation rates outside the reserve, thus concluding that the establishment of the reserve has failed to protect habitat quality and biodiversity. Our case study in Celaque National Park finds the opposite, that the area within the park has experienced limited land cover change when compared to the surrounding landscape. However, there is a significant shift in landscape pattern within the park, that is only apparent between 1996 and 2000, and which could have significant implications for future biodiversity.

This study highlights some of the advances that can be made by incorporating time series and fragmentation analyses from remotely sensed images to analyze the effectiveness of park boundaries. At the same time, analysis of remotely sensed data requires fieldwork to interpret human activities and incentives that relate to land cover change. In this case, fieldwork shows that the boundaries of the park are not well known by the population that lives in and around the park. Except for the requested relocation of approximately 700 people to land outside the park (These locations were requested by the population on learning a park was to be created, Aguilar, 2003), the degree to which the park has compelled people to change land use in the park is not clear. Park guards have minimal training and few resources; the coffee plantations encroaching on the park are legally protected under Honduran law, even within protected areas (Aguilar, 2003; Fonseca et al., 1999). In other parts of the world, the lack of enforcement and inadequate funding has been associated with park degradation (Wells, 1992). In short, our data is robust in showing that the park has been comparatively stable in forest cover as compared to the surrounding landscape. It does not indicate why the park has been effective; topography and lack of roads may constitute more important factors than park boundaries in preserving forest cover.

From previous research relating land cover change to land use in the broader landscape surrounding Celaque (Southworth et al., 2002; Southworth & Tucker, 2001) we can relate the recent change in landscape pattern in the park, to increasing pressure on the land, and concurrent shifts in land use. Increasing agricultural intensification in recent years, through use of mechanized plows and fertilizers, has led to the expansion of pre-existing productive agricultural areas (large patches of deforestation close together). This shift in land use is associated with the abandonment of some less accessible, marginally productive fields, creating small, isolated patches of reforestation. Coffee production may also have an impact, with the increased clearing of large areas required for coffee relating to the increase in size of patches of deforestation in 1996–2000, as compared to smaller patches cleared in previous years for swidden agriculture.

Further, patches of deforestation are smaller than patches of forest/reforestation in 1987–1991 and in 1991–1996. However, by 1996–2000 deforestation patch sizes are significantly larger than forest/reforestation patches in the park. This can again be related to a switch in land use patterns within the study period (agricultural intensification and coffee production) and increasing pressure on the land leading to larger clearings even within the park boundaries. This shift in pattern is not picked up by looking only at the extent of land cover change on each individual date, but needs analysis of pixel level trajectories of change, as is undertaken here. While the amount of change may not be significant, the pattern of change has considerable, potentially negative implications for biodiversity in this species-rich area (Forman, 1995; Haines-Young & Chopping, 1996).

As in several parts of the tropics, there are many complex and interrelated processes driving recent land cover change in the Celaque study area. Agricultural intensification appears related to abandonment of some marginal lands. However, the advent of coffee production affects land cover in previously forested areas. This trend will only increase in the future as the limited areas of mountain coffee production expand. Results emphasize the need for protected areas to interact with local communities, and involve them in management of the park, to ensure effectiveness. This analysis demonstrates the use of change trajectories to measure change quantitatively, to incorporate information on both land cover extent and landscape pattern from satellite image analysis, and to integrate GIS information on the biophysical structure and accessibility of the landscape.

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References

Aguilar, A. (2003). Patterns of forest regeneration in Celaque National Park, Honduras. *Online Journal of Space Communication*, *3*, 31.

Aguilar-Stoen, M., & Dhillion, S. S. (2003). Implementation of the convention on biological diversity in Mesoamerica: environmental and developmental perspectives. *Environmental Conservation*, *30*, 131–138.

Archaga, V. (1998a). Descripción física y caracterización del Parque Nacional Montaña de Celaque. Proyecto Celaque. AFE-COHDEFOR y GTZ. Santa Rosa de Copán. p. 15.

Archaga, V. (1998b). Enfoque de la participación social en el marco del Proyecto Conservación del Parque Nacional Montaña del Celaque. Proyecto Celaque. Santa Rosa de Copán. p. 13.

Bates, D., & Rudel, T. K. (2000). The political ecology of conserving tropical rain forests: a cross-national analysis. *Society and Natural Resources*, *12*, 619–634.

Bishop, K., Phillips, A., & Warren, L. (1995). Protected for ever? Factors shaping the future of protected areas policy. *Land Use Policy*, *12*, 291–305.

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

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

Chavez Borjas, M. (1992). *Cómo Subsisten los Campesinos*. Tegucigalpa: Editorial Guaymuras.

Daily, G. C., Ehrlich, P. R., & Sanchez-Azofeifa, G. A. (2001). Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. *Ecological Applications*, *11*, 1–13.

Fonseca, J. P., Moreno, M. L., & Padgett, G. S. (1999). Estructura florística, uso de recursos y educación ambiental del parque nacional Montaña de Celaque. Tesis Licenciatura en Biologia, Universidad Nacional Autónoma de Honduras, May 1999, p. 104.

Forman, R. T. T. (1995). *Land mosaics: The ecology of landscapes and regions*. Cambridge, UK: Cambridge University Press.

Haines-Young, R., & Chopping, M. (1996). Quantifying landscape structure: a review of landscape indices and their application to forested landscapes. *Progress in Physical Geography*, *20*, 418–445.

Harcourt, C. S., & Sayer, J. (1996). *The conservation atlas of tropical forests: The Americas*. New York: Simon and Schuster.

Hayes, D. J., Sader, S. A., & Schwartz, N. B. (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 Ecology*, *17*, 299–314.

Helmer, E. H. (2000). The landscape ecology of tropical secondary forest in Montane Costa Rica. *Ecosystems*, *3*, 98–114.

Humphrey, C. (2000). *Moon handbooks: Honduras* (pp. 402). Chico, CA: Moon Publications.

Imbernon, J., & Branthomme, A. (2001). Characterization of landscape patterns of deforestation in tropical rain forests. *International Journal of Remote Sensing*, *22*, 1753–1765.

Japan International Cooperation Agency (JICA). (2002). *JICA country profile on environment: Honduras*. Planning and Evaluation Department, Japan International Cooperation Agency.

Jensen, J. R. (2000). *Remote sensing of the environment: An earth resource perspective* (pp. 544). New Jersey: Prentice-Hall.

Kleinn, C., Corrales, L., & Morales, D. (2002). Forest area in Costa Rica: a comparative study of tropical forest cover estimates over time. *Environmental Monitoring and Assessment*, *73*, 17–40.

Liu, J., Linderman, M., Ouyang, Z., An, L., Yang, J., & Zhang, H. (2002). Ecological degradation in protected areas: the case of Woolong Nature Reserve for Giant Pandas. *Science*, *292*, 98–101.

Macleod, R. D., & Congalton, R. G. (1998). A quantitative comparison of change-detection algorithms for monitoring eelgrass from remotely sensed data. *Photogrammetric Engineering and Remote Sensing*, *64*, 207–216.

Mather, A. S., & Needle, C. L. (2000). The relationship of population and forest trends. *The Geographical Journal*, *166*, 2–13.

McGarigal, K., & Marks, B. J. (1994). *Fragstats: Spatial pattern analysis program for quantifying landscape structure, v. 2.0*. Corvallis, OR: Forest Science Lab, Oregon State University.

Mertens, B., & Lambin, E. F. (2000). Land-cover-change trajectories in southern Cameroon. *Annals of the Association of American Geographers*, *90*, 467–495.

Mladenoff, D. J., White, M. A., Pastor, J., & Crow, T. R. (1993). Comparing spatial patterns in unaltered old-growth and disturbed forest landscapes. *Ecological Applications*, *3*(2), 294–306.

Nagendra, H. (2001). Using remote sensing to assess biodiversity. *International Journal of Remote Sensing*, *22*, 2377–2400.

Nagendra, H., Southworth, J., & Tucker, C. (2003). Accessibility as a determinant of landscape transformation in western Honduras: linking pattern and process. *Landscape Ecology*, *18*(2), 141–158.

Parks, S. A., & Harcourt, A. H. (2002). Reserve size, local human density and mammalian extinctions in U.S. protected areas. *Conservation Biology*, *16*, 800–808.

Petit, C., Scudder, T., & Lambin, E. (2001). Quantifying processes of land-cover change by remote sensing: resettlement and rapid land-cover changes in south-eastern Zambia. *International Journal of Remote Sensing*, *22*, 3435–3456.

Pfeffer, M. J., Schelhas, J. W., & Day, L. A. (2001). Forest conservation, value conflict, and interest formation in a Honduran National Park. *Rural Sociology*, *66*, 382–402.

Pineda Portillo, N. (1997). *Geografía de Honduras. Tercera Edición*. Tegucigalpa: Editorial Guaymuras.

Riitters, K., Wickham, J., O'Neill, R., Jones, B., & Smith, E. (2000). Global-scale patterns of forest fragmentation. Conservation Ecology 4, 3. Available: URL: http://www.consecol.org/vol4/iss2/art3.

Rivard, D. H., Poitevin, J., Plasse, D., Carleton, M., & Currie, D. J. (2000). Changing species richness and composition in Canadian national parks. *Conservation Biology*, *14*, 1099–1109.

Rivas, R. (1994). *Situación de pueblos Indígenas de Honduras* (p. 495). Tegucigalpa: Guaymuras.

Sader, S. A. (1995). Spatial and temporal forest change in the Northern Peten Region of Guatemala. *Photogrammetric Engineering and Remote Sensing*, *61*(9), 1145–1151.

Sader, S. A., Hayes, D. J., Hepinstall, J. A., Coan, M., & Soza, C. (2001). Forest change monitoring of a remote biosphere reserve. *International Journal of Remote Sensing*, *22*, 1937–1950.

Sanchez-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.

Sanchez-Azofeifa, G. A., Quesada-Ameto, C., Gonzalez-Quesada, P., Dayanandan, S., & Bawa, K. S. (1999). Protected areas and conservation of biodiversity in the tropics. *Conservation Biology*, *13*, 407–411.

Sanchez-Azofeifa, G. A., Rivard, B., Calvo, J., & Moorthy, I. (2002). Dynamics of tropical deforestation around national parks: remote sensing of forest change on the Osa Peninsula of Costa Rica. *Mountain Research and Development*, *22*, 352–358.

Schelhas, J. (1991). A methodology for assessment of external issues facing national parks, with an application in Costa Rica. *Environmental Conservation*, *18*, 323–330.

Schelhas, J. (1996). Land use choice and change: intensification and diversification in the lowland tropics of Costa Rica. *Human Organization*, *55*, 298–306.

Schwartz, M. W. (1999). Choosing the appropriate scale of reserves for conservation. *Annual Review of Ecology and Systematics*, *30*, 83–108.

Sokal, R. R., & Rohlf, F. J. (1981). *Introduction to biostatistics*. (2nd ed.). Washington, DC: Island Press.

Southworth, J., Munroe, D., & Nagendra, H. (2004). Land cover change and landscape fragmentation—comparing the utility of continuous and discrete analyses for a western Honduras region. *Agriculture, Ecosystems and Environment*, *101*(2–3), 185–205.

Southworth, J., Nagendra, H., & Tucker, C. M. (2002). Fragmentation of a landscape: incorporating landscape metrics into satellite analyses of land cover change. *Landscape Research*, *27*, 253–269.

Southworth, J., & Tucker, C. M. (2001). Forest cover change in western Honduras: the role of socioeconomic and biophysical factors, local institutions, and land tenure. *Mountain Research and Development*, *21*, 276–283.

Thacher, T., Lee, D., & Schelhas, J. (1997). Farmer participation in reforestation incentive programs in Costa Rica. *American Journal of Agricultural Economics*, *79*, 1704–1704.

Thomlinson, J. R., Bolstad, P. V., & Cohen, W. B. (1999). Coordinating methodologies for scaling landcover classifications from site-specific to global: steps towards validating global map products. *Remote Sensing of the Environment*, *70*, 16–28.

Tucker, C. M. (1999a). Private vs. communal forests: forest conditions and tenure in a Honduran community. *Human Ecology*, *27*, 201–230.

Tucker, C. M. (1999b). Evaluating a common property institution: design principles and forest management in a Honduran community. *Praxis*, *15*, 47–76.

Uebelhör, K. (1998). *El manejo de la zona de amortiguamiento del parque Nacional Montaña de Celaque: Consideraciones preliminariesDocumento Técnico No. 3.* Tegucigalpa: Corporación Honduras-Alemania Programa Social Forestal (PROFOR).

Wells, M. (1992). Biodiversity conservation, affluence and poverty: mismatched costs and benefits and efforts to remedy them. *Ambio*, *21*, 237–243.

World Bank (2001). Honduras poverty diagnostic 2000. Report #20531-HO. Latin American and Caribbean Region: Poverty Reduction and Economic Management Unit, World Bank.