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Are conservation strategies effective in avoiding the deforestation of the Colombian Guyana Shield?

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A B S T R A C T

There is general agreement regarding the importance of analysing the territories' roles under different biodiversity management figures in order to support better decision making in the management of natural resources in tropical countries. In this study we analyse the deforestation process to address the question of whether existing strategies such as national protected areas (PAs) and indigenous reservations (IRs) are effective protecting forests in the Colombian Guyana shield. We analyse whether these territories have successfully halted deforestation and agricultural frontier expansion by comparing deforestation occurring within these areas with their surroundings from 1985 to 2002. We also evaluate the impact of roads, illicit crops, and the size of PAs and IRs on deforestation rates. The results indicate that deforestation levels along the outside borders of both management figures were almost four times higher than inside declared PAs and 1.5 times higher than in IRs. However, within IRs, the loss of forested ecosystems was approximately six times greater than inside national parks. As a whole, roads were a significant factor associated with the changes in the region, as well as the influential expansion of coca cultivation particularly outside the national parks. The size of the PAs and indigenous lands also determined their positive impact as barrier against deforestation. Our results suggest strong pressure on areas surrounding PAs, driven by economic forces such as illegal crop expansion, particularly in the last decade. Indigenous lands with small territories have suffered intensive deforestation processes since the 1980s, but changes have been less dramatic in larger areas. Today, PAs are an effective barrier to deforestation, especially given their large extension, but are still under high risk. Future management plans should consider a designed infrastructure development paired with the establishment of new indigenous reservations with minimum viable sizes in order to control accessibility, natural resources extraction, and deforestation.

1. Introduction

The conversion and degradation of forests threaten the integrity of forested ecosystems worldwide (Nepstad et al., 1999; Gascon et al., 2000; Achard et al., 2002). In particular, tropical forests play an important role in preserving many ecosystem services and are the primary focus of many conservation efforts, because they contain some of the most species-rich and highly threatened habitats in the world (Myers et al., 2000). Deforestation patterns vary across regions. South America is one of the planet's regions containing larger blocks of forests, with most forest area per capita and fewer fragmented forests, partly as a result of their inaccessible (and thus unexploitable) locations (Rudel, 2006). Indeed, roads and other agents of change such as small-scale farmers, shifting cultivators or population growth have traditionally been associated with tropical deforestation (Rudel, 2006; Butler and Laurance, 2008). Today a shift away from deforestation towards a more industrially driven process is beginning to appear in some regions (Rudel, 2007; Butler and Laurance, 2008). The increasingly deforested frontiers of tropical forests and agricultural expansion has resulted in more focused attention on the best approaches for conservation and management of protected areas, as well as the development of other strategies for biodiversity conservation, such as the role of indigenous reserves (Foster et al., 1999; Du Toit et al., 2004; Román-Cuesta and Martínez-Vilalta, 2006; Nepstad et al., 2006; Oliveira et al., 2007). These areas are central to conservation strategies because they are designed to safeguard remaining habitats and species (DeFries et al., 2005; Joppa et al., 2008). However, when feasible, conservation efforts have tended to focus on the creation of new areas in remote or low density populated areas (Rudel, 2006).

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Evaluating the effectiveness of PAs is difficult, especially given the limited data on ecological and social conditions and their changes over time (Naughton-Treves et al., 2005). Measurements of effective long-term protection of biodiversity in PAs have usually been proposed under broad terms (Hockings et al., 2000). Some studies have even developed a methodology to quantify this effectiveness by using questionnaires on aspects related to human pressure and management activities (Bruner et al., 2001). These assessments give a general picture of the conservation and management of tropical biodiversity (Bruner et al., 2001; Rodrigues et al., 2004; DeFries et al., 2005), often revealing threats to national parks caused by clearing, hunting and logging, however, the assessments also generally show these problems to be less severe inside parks than in surroundings (Bruner et al., 2001; Naughton-Treves et al., 2005). It should be noted, however, that these kind of global assessments are often driven by data availability or ease of data collection on a regional or global scale, meaning that there are areas of the world which remain understudied.

The Amazon basin, the Brazilian portion, in particular, contains the world's highest absolute rate of deforestation (Laurance et al., 2001), and has been a primary focus in debates between conservation and development, as well as the effectiveness of conservation units in the whole watershed (Cardille and Foley, 2003; Chomitz and Thomas, 2003; Fearnside, 2005; Joppa et al., 2008). It is also one of the regions where large industrially driven deforestation trends are observed either by agriculture, ranching or oil and gas development (Butler and Laurance, 2008). In contrast, the Guyana shield – another large area of tropical forest wilderness in South America – has the lowest deforestation rate in the world, with almost 90% of its territory in a pristine state (Ter Steege et al., 2000). While there is still no presence of major industrial logging, mining, or agricultural activities in the Guyana shield, the area still faces increasing threats, such as colonisation or increased mining activities (Ter Steege et al., 2000). Little attention has been paid to the extent and drivers of deforestation in this region, especially inside and beyond the boundaries of conservation units, there has also been little attention paid to the role of illicit crops and the presence of indigenous populations with a long-time presence in the region. This lack of information is particularly evident at the Colombian national level. Colombia currently houses nearly 49,000,000 ha of tropical lowland, montane and dry forests (Etter, 1998), 80% of which is nominally protected in natural parks and indigenous or forest reservations (Ponce, 1999). These forests are conservation priorities because they are the last repositories of a highly diverse and endemic biota (Myers et al., 2000). However, despite previous government attempts to manage colonisation processes, rapid deforestation remains virtually uncontrolled (Armenteras et al., 2006; Etter et al., 2005, 2006). In tropical lowlands this is mainly due to traditional drivers such as high colonisation pressures and the intensification of illegal coca (*Erythroxylum coca*) crops (Viña et al., 2004). Despite future government development plans that include large-scale agriculture and oil and gas development that might lead to similar trends occurring in the continent's other tropical forests (Soares-Filho et al., 2006; Butler and Laurance, 2008), the region still remains under a traditional shifting agriculture, cattle ranching and low population densities – factors that favour the cultivation of illegal crops in marginal lands, decreased accessibility, little institutional presence or law enforcement, and the occasional presence of illegal or armed groups (Armenteras et al., 2006).

This study builds on previous studies that have analysed satellite imagery of deforestation in and around wilderness PAs (Nepstad et al., 2006; Joppa et al., 2008) and further explores issues of addressing deforestation in areas through the inclusion of a previously unanalysed area, and the consideration of significant but little studied issues, such as illegal crop production and the presence of indigenous reserves in the Guyana shield. In this study we also analyse how effective protected areas and indigenous territories have been at mitigating deforestation within their boundaries, as compared with their adjacent buffers (defined as concentric areas surrounding the boundaries of the protected area, whose final area equals the total land of each protected area; see Román-Cuesta and Martínez-Vilalta, 2006) throughout the Colombian Guyana shield. We use a GIS database and satellite data, we examine the extent of existing natural forests, as well as deforestation rates for 1985, 1992 and 2002 within and surrounding PAs and indigenous reservations. Some conservation scientists are increasingly convinced that indigenous residents are necessary actors for the long-term conservation of tropical forests (Schwartzman et al., 2000; Schwartzman and Zimmerman, 2005; Nepstad et al., 2006), and that traditional forest management practices of these indigenous populations can eventually help maintain the natural and cultural values of a region. Thus, we compare the effectiveness of uninhabited (national parks) and inhabited (national indigenous reservations and indigenous reservations) protected territories to mitigate the expansion of the agricultural frontier. Several authors have noted the relationship between site accessibility to markets (through roads or rivers) and the presence of deforestation in lowland tropical forests (Barros Ferraz et al., 2005; Kirby et al., 2006; Mas, 2005; Oliveira et al., 2007). Infrastructures favour land occupation and, illegal activities (such as coca growing), and thus support legal or illegal resource extraction, access to markets, degradation of forests and the fragmentation and deforestation of natural forests. As mentioned, illegal activities in Colombia – especially in remote areas – are also related to armed conflicts and population displacement (Davalos, 2001; Etter et al., 2005), which indirectly affect the expansion of the agricultural frontier, in some cases, land abandonment resulting from these activities cause an increase in secondary and transformed ecosystems, which is highly disturbing to tropical forests. In this study, we evaluate whether management and conservation areas in the Colombian Guyana shield have fewer changes in land cover than unprotected neighbouring areas in the region, we also analyse the role of other driving factors, such as reservation size, roads and the presence of illegal crops.

2. Methods

2.1. Study area

Colombia, the fourth largest country in South America, has a population of nearly 1.4 million and is home to some 84 different ethnic groups (Dane, 2005 Census). These ethnic groups make up 3.3% of the country's total population. They are primarily located in rural areas spanning 718 different indigenous reservations. Colombia is considered one of the world's richest countries in terms of both biological and cultural diversity. The National Natural Parks System consists of 53 natural areas, covering about 10% of the national territory.

The Guyana shield region in South America covers approximately 2.5 million km^2 (Fig. 1). This region generally bordered in the south by the Amazon river, the Japurá-Caquetá river in the southwest, the Sierra of Macarena and Chiribiquete in the west, the Orinoco and Guaviare rivers in the north and northwest, and the Atlantic Ocean in the east (GSI, 2002). Vegetation types found in the region include sandstone Tepuis (or Table Mountains), white sand vegetation, large savannah areas, coastal swamp forests; gallery forests, and several tropical rain forest systems. The Colombia Guyana Shield, situated between the Amazon and Orinoco basins, is a territory stretching over 200,000 km^2 and belongs to the Guyana Western province of the Guyana phytogeographic region (Huber, 1994; Berry et al., 1995). It is comprised mainly of rocky outcrops, sierras and isolated mountains (with a maximum

Fig. 1. Map locating national protected areas and indigenous reservations studied in the Colombian Guyana Shield.

altitude of 1500 m) and has a climate ranging from dry to hot, tropical and humid. Due to its location, the region is high in biodiversity and endemism, but human pressures have progressively expanded agricultural frontiers to the area. Economic activities in the region are mainly related to the extraction of natural resources, followed by the establishment of pastures and crops. Despite (or because of) the lack of proper infrastructure and difficult physical access, illegal crops (mainly coca) are widely present in the region. Nearly 22% of the recorded coca crops in 2004 were located in our study area (UNODC, 2006). According to national census data, the region's total non-indigenous population is 166,230 (Sisben, 2003–2004), 55% of which are located in towns and small settlements and 45% of which are located in rural areas. The indigenous census officially reports a population of 32,764 (Arango and Sánchez, 2004), 58% of which live in the Tuparro National Park area, followed by 30% in the Puinawai area. Of the ethnicities of the indigenous peoples within the studied area, 8% are Nukak and 4% are Chiribiquete. The Macarena area currently does not have any registered indigenous reservations (Rodríguez et al., 2006).

The study area includes five watersheds of the northern region of the northern Colombian Amazon region (La Primavera, Duda, Alto and Bajo Inirida and Mesay), comprising a total area of 12,611,760 ha. This area contains three national parks (Sierra de la Macarena, Serranía de Chiribiquete and Tuparro), two national natural reservations (Nukak and Puinawai) and 44 indigenous reservations (Appendix A). The most representative biomes of the study area are the tropical forests of Amazonia and Orinoco and the pedobiomes (areas with extreme soil types and azonal vegetation; Walter, 2002) of the humid tropical Amazonia and Orinoquia zonobiome. Natural ecosystems cover a total area of 11,728 936 ha (93.2% of the total area). The areas of greatest transformation are the Nukak and Macarena areas which account for another 882,823 ha (6.8% of the area). The Tuparro and Chiribiquete areas contain the greatest coverage of remaining natural ecosystem, accounting for more than 97% of their catchment area, while the Macarena region has the lowest percentage of natural ecosystems (84%). In general, natural ecosystems have transformed into pastures and a pasture–crop matrix in the Nukak and Macarena areas. The other regions contain an assortment of small traditional crops (e.g. small-scale agriculture as practiced by the indigenous population, known locally as "*chagras*").

2.2. Methodology

Geographic information was collected from LANDSAT TM and ETM satellite images (Rodríguez et al., 2006). We carried out digital classifications to identify land cover changes between 1985 and 2002 at a spatial resolution of 30 m. We selected three time periods for the analysis – 80–90s represented by the years 1985–1992; 90–00s, represented by the years 1992–2002, and 80–00s, represented by the years 1985–2002. Land cover classification was undertaken using ERDAS Imagine V8.7 (Leica Geosystems, 2005). The Error Matrix, Accuracy Totals, and Kappa Statistics were used to gauge the accuracy of the classification and protocols similar to those used by Meidinger (2003) were used to evaluate the quality of the map classification (employing field work and aerial photography). The final map presented an overall accuracy of 95% for polygons bigger than 25 ha (Rodríguez et al., 2006). A digital database of the road network with a scale of 1:250.000 was obtained from a previous project (Romero et al., 2004) and was updated with the aforementioned satellite image digital processing.

Using this data, we built a GIS database and classified land covers into the following categories: (i) natural, including tropical rainforests, gallery forests, tree vegetation and natural savannas with no detected disturbances, (ii) transformed ecosystems, mainly including agricultural systems, crops, pastures and a matrix of some urban and settlement areas, (iii) secondary ecosystems, including secondary forests, secondary vegetation, disturbed forests primarily caused by pasture and agriculturally abandoned lands (as a transition state between the first two categories) and disturbed forests caused by logging or similar activities (Etter et al., 2005), (iv) rivers; and (v) roads. Information regarding illicit crops in the region was obtained from the global illicit crop monitoring program of the United Nations Office on Drugs and Crime (UNODC, 2006), represented in Colombia by the Integrated System for Illicit Crops Monitoring project or SIMCI (*Sistema Integral de Monitoreo de Cultivos Illicitos*). This system has consistently monitored illicit crops in the region since 2000 and has provided access to coca survey data for the study area. We use the GIS package ArcGIS (ESRI) to conduct all digital spatial analysis.

To compare land cover changes inside and along the national PAs or indigenous reservations, we derived 10 km buffer zones around their perimeters. The size of this 10 km buffer around the areas was chosen to allow comparison with other effectiveness studies (Bruner et al., 2001; Sánchez-Azofeifa et al., 2003; Román-Cuesta and Martínez-Vilalta, 2006) and also because this particular distance provides similar environmental conditions, avoiding heterogeneity in spatial variables that could otherwise bias the assessment (Mas, 2005). For the analysis we excluded reservations with small areas inside the study area (<5000 ha) and also grouped adjacent reservations (creating contiguous indigenous areas), thus avoiding superimposed buffers for those territories. Based on these criteria, we were able to quantify change rates for five PAs, and 14 indigenous territories, representing a total of 8,196,028 ha under some type of conservation management. We also created a 10 km road buffer dataset, and calculated the loss of natural ecosystems within this buffer area. We then overlaid this layer with the PAs layer to determining the percentage of area loss around roads which occurred inside PAs, as a way to control and relate it to accessibility.

We used the term "deforestation" to refer to losses of natural ecosystems. Change rates were calculated using R (as in Fearnside, 1993), but rates were standardised by the total analysed area in order to avoid undetermined values due to zeros, especially in secondary and transformed ecosystems to which R was also applied. The formula applied used R as $R = (A_1 - A_2)/(A_{ta} * (t_2 - t_1))$ where A_1 and A_2 were the areas in hectares at years 1 and 2 two respectively (e.g., if looking at the period 1985–1992, A_1 and A_2 were the forest covers in 1985 and 1992, respectively), and A_{ta} was the total area analysed for that specific reservation, and t was time in years.

We used non-parametric statistics because most variables were not normally distributed. We first applied Wilcoxon matched paired tests to compare change rates inside and outside buffer areas for both national parks and indigenous reservations. We also performed the Mann Whitney test to compare PAs and indigenous reservations. Using Spearman correlations, we analysed the relationship between ecosystem change rates and the following explanatory variables: length of roads, area of illicit crops, and the size of the management area (another factor that may have implications in future conservation design). These explanatory variables were not highly correlated between them: the only significant correlation was in the elements of illicit crops area and road length s ($R^2 = 0.22$, $p = 0.0016$). Given the low number of samples, all significance tests were carried out at the $p = 0.10$ level. We used SPSS software for all statistical analysis.

3. Results

3.1. Deforestation

Natural ecosystems still dominate the entire study area, but there was nonetheless an annual average deforestation rate of 0.16% over the period analysed. Regionally, our results indicate that the loss of natural ecosystems in absolute numbers throughout the 17 years was subtle in the studied area (419,243 ha) in comparison to other global tropical regions, but our results also confirm changes in South America were more evident in areas around indigenous reservations and PAs rather than inside these areas. Both protected areas (45,739 ha) and indigenous reservations (35,891 ha) have lost less than 1% of natural forests present in the 80s, while their buffer areas follow a pattern of 5–7% loss (Fig. 2). Inside national parks, the following natural ecosystems were reduced: Macarena by 17 936 ha; Puinawai by 9715 ha; Nukak by 8727 ha; Tuparro by 5774 ha; and Chiribiquete by 3584 ha. These results represent a small percentage of total deforestation (10% for protected areas, 8% for indigenous reserves) occurring in the region. Mean deforestation rate results for indigenous reservations were 5.8 times higher than those in national protected areas. Despite these results, only a few significant differences in deforestation were found between these two management figures for the 90s–00s period (Table 1).

On the other hand, significant differences in deforestation rates were found inside national PAs versus their buffer areas (Table 1), with relatively higher deforestation rates outside PAs in the 90–00s decade than in the 80–90s decade. On average, deforestation from the 80–00s was 3.98 times higher outside the borders of national PAs (0.28 per year) than inside them (0.071 per year) (Fig. 3). Deforestation rates were 1.49 times higher outside indigenous reservations than inside them (Fig. 3), but this difference was only significant for the 80–90s period.

A similar pattern was observed inside and outside PAs and reserves as a result of more extensive land cover change between secondary and transformed ecosystems. The transformation of ecosystems as a whole increased everywhere (Fig. 2b), from 6474 ha to approximately 33,285 ha in PAs and 27,7600 ha more inside indigenous reservations. Expansion in the buffers was at least four times the area that was present in the 80s, with an observed increase of area of 74% and 84% around protected areas and reserva-

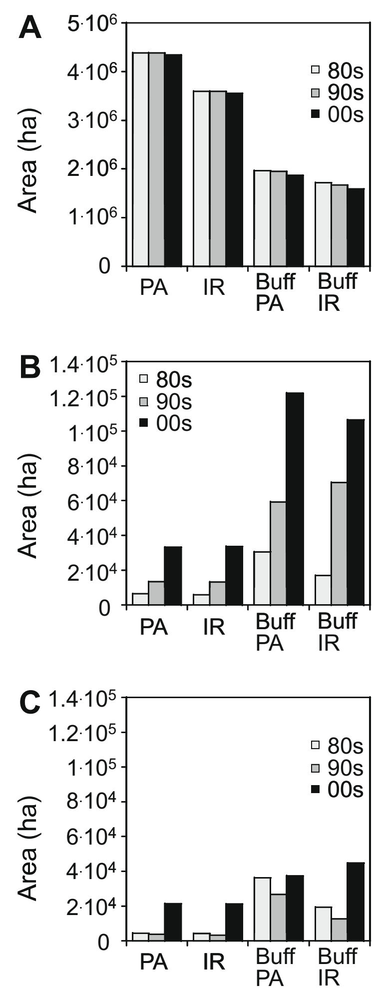

Fig. 2. Hectares of (A) natural, (B) transformed and (C) secondary ecosystems in national protected areas and indigenous reservations and their buffers in the 80s, 90s and 00s. Note the different scale used in the Y-axis of the three graphs. PA, protected area; IR, indigenous reserve; Buff PA, buffer of protected area; Buff IR, buffer of indigenous reserve.

Table 1
Results of the non-parametric test carried out for the comparison of change rates of the different ecosystem types (NE, natural ecosystems; TE, transformed ecosystems; SE, secondary ecosystems), road length and coca cultures between the two management categories and between each category and their buffers (PA, protected area; IR, indigenous reservation; Buff PA, buffer of the protected area; Buff IR, buffer of the indigenous reservation) in the three periods considered (80–00, 80–90 and 90–00). ns, not significant.

Non parametric tests between categories of management	NE			TE			SE			Roads	Coca
	80–00	80–90	90–00	80–00	80–90	90–00	80–00	80–90	90–00		
PA and IR[a]	ns	ns	$Z = 2.2$, $p = 0.026$	ns	ns	ns	ns	ns	ns	ns	Z = 1.9, p = 0.058
PA and Buff PA[b]	$Z = 2.0$, $p = 0.043$	ns	$Z = 2.0$, $p = 0.043$	$Z = 2.0$, $p = 0.043$	ns	$Z = 1.8$, $p = 0.068$	ns	ns	ns	Z = 1.7, p = 0.079	Z = 1.8, p = 0.06
IR and Buff IR[b]	ns	$Z = 1.7$, $p = 0.080$	ns	$Z = 2.0$, $p = 0.041$	$Z = 2.0$, $p = 0.046$	ns	ns	ns	ns	Z = 2.5, p = 0.013	ns

[a] Statistical test: Mann Whitney test.
[b] Statistical test: Wilcoxon matched pairs test.

tions, respectively. The area in secondary forests (Fig. 2c) was higher in the 80s around PAs (36,203 ha) than in indigenous reservations (19,464 ha), but the situation was reversed in the 00s (37,533 ha around national parks in contrast to 44,681 ha in the buffer of indigenous reservations).

The rate of change of transformed ecosystems in the 80–90s period was significantly higher in the buffer of PAs than inside them, and there was a similar significant increase in transformed ecosystems around indigenous reservations as opposed to the Pas inside them (Table 1). As with deforestation rates, the decades of highest change in transformed ecosystems did not coincide between national PAs and indigenous reservations. Secondary ecosystems showed no significant differences among categories of management or among any analysed time period (Table 1).

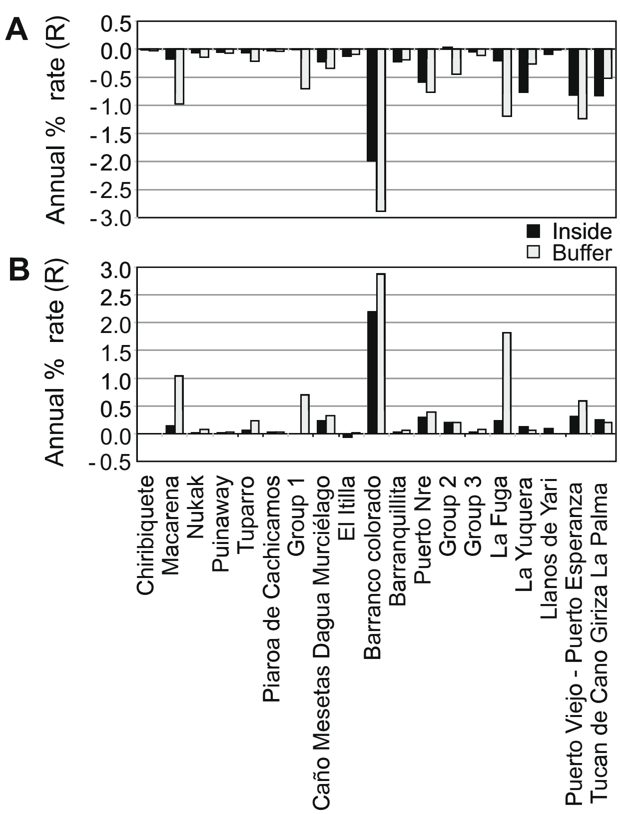

Fig. 3. Change rate for (A) natural ecosystems and (B) transformed ecosystems in each national protected area and indigenous reservation (in black) and its buffer (in grey) from 1985 to 2002. PA, protected area; IR, indigenous reserve; Buff PA, buffer of protected area; Buff IR, buffer of indigenous reserve. The indigenous reservations included in Groups 1, 2–3 are indicated in Appendix A.

3.2. Drivers of change

The three variables (roads, coca crops and size of the management area), showed significant relationships with the different types of ecosystem management category (Table 1). There was no significant difference in the length of roads in PAs compared to indigenous reservations (Table 1). Road length was significantly greater (by at least three times) in the buffer outside both PAs and indigenous reservations than inside them (Table 1). In both cases, when management areas had road infrastructure developments of any kind, the areas were more likely to have forest loss than those without accessibility. Indeed, roads were also a strong factor in land cover changes occurring outside the borders of both management categories, the greater the number of road kilometers, the greater the land cover change figures. The results of the analysis of the 10 km buffer around roads showed that of the total loss of natural ecosystems reported in the study area, 336,347 ha occurred within this buffer (80% of total documented deforestation). Of these losses, only 15,526 ha (representing 3.7% of the total deforested study area) were inside PAs and 5400 ha (1.3% of the total deforested study) were inside indigenous reserves. This means that only 33% of deforestation inside the PAs (45,739 ha) and only 15% of deforestation inside indigenous reserves (35,891 ha) could be explained by accessibility by roads. As expected, most of this activity occured in two areas: the Macarena National Park (12,262 ha) and Tuparro (2305 ha).

In the case of illicit crops, results showed larger areas containing coca crops within indigenous reservations than within national parks (Table 1). There was also a significant difference between the hectares of coca grown inside and outside national PAs. Illegal crops were also specifically related to deforestation and land cover changes in the 10 km buffers outside both PAs and indigenous reservations. The difference between indigenous reservations and their buffers carried no substantial significance (Table 1), although the area of coca outside reservations was on average 3 times higher than areas inside them. Our results also indicate that during the studied time period, the size of the management area (i.e., a national PA or an indigenous reservation) had a strong negative correlation to deforestation, both inside their limits and along their buffers (Table 2); the larger the management area, the lower the loss rate.

4. Discussion

In the Colombia Guyana shield, national protected areas have slower deforestation rates and are better at slowing deforestation rates than indigenous reservations. As concluded in the case of the Peruvian Amazon (Oliveira et al., 2007), our results suggest that both management types can be an effective way of protecting forests.

Table 2
Spearman correlation analyses between change rates of the different ecosystem types (NE, natural ecosystems; TE, transformed ecosystems; SE, secondary ecosystems) and the different factors affecting them (coca cultivation, roads and area size) for the periods 80–00s and 90–00s and the different categories of management (PA, protected area; Buff PA, buffer of the protected area; IR, indigenous reservation; Buff IR, buffer of the indigenous reservation). In each case, the rho value and its significance are indicated. ns, not significant.

Category	Coca crops						Roads						Area size					
	NE		TE		SE		NE		TE		SE		NE		TE		SE	
	8000	9000	8000	9000	8000	9000	8000	9000	8000	9000	8000	9000	8000	9000	8000	9000	8000	9000
PA and Buff PA[a]	ns	0.929 0.001	ns	−0.905 0.002	ns	ns	ns	ns	ns	ns	ns	ns	−0.763 0.010	−0.927 0.001	0.837 0.003	ns	ns	ns
IR and Buff IR[b]	ns	ns	ns	ns	0.588 0.001	ns	ns	ns	ns	0.582 0.004	ns	ns	−0.518 0.005	−0.460 0.031	ns	ns	ns	ns
Buff PA and Buff IR[c]	ns	0.646 0.009	ns	ns	0.631 0.004	0.719 0.003	0.668 0.002	0.821 0.000	0.559 0.013	−0.818 0.000	ns	ns	−0.472 0.041	ns	ns	ns	ns	ns
PA and IR[c]	ns	ns	ns	0.518 0.048	ns	ns	ns	ns	ns	.538 .038	ns	ns	−0.667 0.002	−0.768 0.001	0.507 0.027	ns	ns	0.618 0.014

[a] $n = 10$.
[b] $n = 28$.
[c] $n = 19$.

However, legally established national PAs are less likely to be affected by a colonisation wave than indigenous reservations. Despite this fact, protected areas have been under increasingly stronger pressures in their surroundings since the 90s. There are clear differences between the performance of PAs versus that of indigenous reserves, and most of the variability can be explained by the history of colonisation and the area's proximity and accessibility to the Andes. The Andes houses most of the Colombian population and is the origination of the colonisation waves. The general relation between deforestation rates and coca crops for indigenous reservations is strong, especially in the 80s, when armed conflicts related to illegal activities forced widespread land abandonment and directed human migration towards small indigenous reservations that had been previously established close to the colonisation front. In the 90s coca fields might have extended to the outskirts of some of the national PAs (Macarena, Nukak and Puinawai), all of which were located near colonisation fronts. In fact, the Macarena National Park (the closest park to the Andes), is the most transformed of the five protected areas in this study, showing higher rates of deforestation, more hectares of coca crops, and more roads. This can be explained by long-term colonist exploitation that has occurred since the 70s (Armenteras et al., 2006). This is mainly related to cattle farming and illegal crops, which stem from weak government policies of the 80s and 90s, which led to the presence of multiple illegal armed groups. No indigenous lands have been designated in the area. Meanwhile, Chiribiquete National Park is found in a highly inaccessible area, is nearly without any transport infrastructure, and has some of the lowest deforestation rates of any national PAs. Puinawai and Nukak follow closely, both are national PAs but are managed by indigenous peoples, and are also difficult to access. Finally, Tuparro National Park, which is close to the Venezuelan border, has the second highest land cover change of all national PAs, despite being highly inaccessible from within Colombian territory. Neither Chiribiquete nor Tuparro have been widely opened to illicit crop expansion. Conversely, Nukak and Puinawai contain the presence of coca crops both inside and in their buffers, though not at the same levels as Macarena. This can be partly explained by Puinawai, but cannot fully account for the indigenous populations' use of coca for traditional purposes. In the case of Nukak, recent facts suggest a forced migration of indigenous population by illicit groups involved in illegal coca traffic.

While Brazilian indigenous lands have been proven to reduce deforestation in active agricultural frontiers in the Amazon (Nepstad et al., 2006), indigenous lands in the Colombian Guyana shield have not been as successful at reducing deforestation as have national PAs. The Colombian indigenous territories play an important role in diminishing deforestation rates within their borders, but our results still suggest that they have high rates of deforestation along their 10 km buffer zones. Most of this deforestation occurred in the 80s with the coca boom and resulting migration of colonists, but there is much variability between individual reservations. These variables can be partly determined by the size of the territories, but are also related to their accessibility and the individual colonisation history of the area. We found significant correlation between deforestation rates and the size of the indigenous reservations, smaller reservations are clearly less able to inhibit deforestation along their borders or even inside them (Fig. 3). However, some of the 'hotspots' of deforestation in indigenous reservations were located in the Department of Guaviare along the colonised border of San José del Guaviare (a former peace zone). Two of the smallest reservations, Barranco Colorado and La Fuga, are located in a region that is easily accessed, containing the highest density of road kilometers (after the Macarena area). These two reservations are also located on roads built following a wave of forest exploitation by non-indigenous populations, mainly related to cattle farming. The area's recent river transportation network is related to the illegal extraction of resources and the expansion of coca crops. Other deforestation 'hotspots' in or around indigenous lands are found in the same department but along the Miraflores-Barranquillita area south of the previously mentioned colonisation frontier. Miraflores is a municipal level settlement with mostly indigenous inhabitants, which also has a floating population that fluctuates with the economic and productivity cycles of the coca "boom". As a result, some reservations presented higher deforestation rates inside the reservations than outside, which can be partly explained by the major deforestation that had already occurred outside their limits. On the other hand, the lowest deforestation rates for indigenous lands are present in the reservations around the Tuparro National Park and Puinawai area, these are coincidently the most inaccessible areas, made up of mostly indigenous populations, with small settlements of colonist families that dedicate themselves to the illegal trafficking of contraband goods across the border. However, due to lack of data, forest resource extraction activities such as legal or illegal logging, poaching or hunting could not be considered in this study.

Transformation of forests lead to an increase in transformed ecosystems in most of the areas analysed for deforestation. The increase in this type of ecosystem within the study area between 80s–00s occurred much less intensely inside protected areas and indigenous reserves than around their buffers. This makes sense, since land cover change from forests to other land cover (driven mainly by agricultural activities), damages forests, removing forest cover through the extraction of resources, clear cutting, and the establishment of settlements (Etter et al., 2006; Rodríguez et al., 2006). An increase in transformed forests for these ecosystems is also, as expected, related to the road network around reservations and parks. Conversely, our results on secondary ecosystems (used as a transition stage between natural and transformed ecosystems), were related more to hectares of coca crops. Again, the increase of secondary ecosystems since the 80s in the region occurred preferentially in the buffers of protected areas or indigenous reserves than inside them. This may suggest that there has been an important abandonment of coca fields, or a migration of crops to other areas as a result of government eradication programs. However, it seems they may not be reducing production, but rather expanding cultivation zones. This is particularly important since secondary ecosystems showed no relation at all to existing roads.

Following Joppa et al. (2008)'s classification of *de facto* versus *de jure* protection, areas in the Guyana shield probably contain a combination of both protection types. Some areas escape human activity due to physical inaccessibility; others are probably protected (and some only partially, as in cases such as Macarena) because of their legal status. Neither the five protected areas nor most of the reservations are in fact *paper parks* (Joppa et al., 2008), but some of the indigenous reservations in the area can nonetheless be labelled as such, since they do not provide conservation or sustainable use of forests. Future conservation and development paths (both regional and international), including policies, legislation and land cover planning must consider spatial planning as an important component of successful development. Successful development must balance the economic, cultural, social and environmental aspects of the relatively well-preserved Guyana Shield in Colombia. In agreement with Joppa et al. (2008), the government, its managers and its decision-makers should take into serious consideration, along with biological and cultural criteria, the remoteness and size of future parks and, reservations, and should even consider the expansion of existing management areas to increase their extent.

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Appendix A

Main characteristics of the national protected areas (PA) and indigenous reservations (IR) analysed in this study including deforestation rates inside and along their buffers. N/A, information not available.

Name	Protection Category	Population	Total area (ha)	Area inside the study area (ha)	Deforestation rate inside the area (1985–2002)	Deforestation rate in the 10-km buffer (1985–2002)
Chiribiquete	PA	0	1,303,829	1,303,829	0.0162	0.0283
Macarena	PA	0	628,052	628,052	0.1695	0.9724
Nukak	PA	0	874,567	874,567	0.0590	0.1418
Puinaway	PA	0	1,115,456	1,115,456	0.0518	0.0707
Tuparro	PA	0	554,401	554,401	0.0620	0.2135
Barranco Colorado	IR	157	9327	8353	1.9901	2.8879
Barranquillita	IR	191	22,265	22,265	0.2184	0.1887
Cano Mesetas-Dagua y Murcielago	IR	99	83,720	83,720	0.2247	0.3396
El itilla	IR	44	8719	8719	0.1265	0.0890
Group 1 (includes Cano Bachaco Guaripa, La Hormiga y Guacamayas Maipore	IR	279	35,385	34,252	0.2072	1.1916
Group 2 (includes Lagos del Dorado, Lagos del paso, Bacat-Arara, Vuelta del Alivio, Yabilla II)	IR	1458	377,530	148,413	0.7644	0.2599

(*continued on next page*)

Appendix A (*continued*)

Name	Protection Category	Population	Total area (ha)	Area inside the study area (ha)	Deforestation rate inside the area (1985–2002)	Deforestation rate in the 10-km buffer (1985–2002)
Group 3 (includes Barranco Ceiba y Lag., Cano Jabon, Cuenca Media y alta del rio Inirida, Nukak Maku, Parte alta del rio Guainia, Remanso Chorro Bocon, Rios Cuiari e Isana, Tonina-Sejal-San Jose)	IR	5958	6,081,660	3,202,269	0.0933	0.0122
La Fuga	IR	145	8360	6215	0.0254	0.0367
La Sal	IR	191	3275	20,866	0.5839	0.7648
Llanos de Yari (Yaguara II)	IR	196	146,500	91,300	0.8127	1.2362
Piaroa de Cachicamo	IR	N/A	16,562	16,562	0.8227	0.5150
Puerto Nare	IR	116	23,368	23,071	0.0041	0.6974
Puerto Viejo y puerto Esperanza	IR	117	9100	8973	−0.0349	0.4430
Tucan de Caño Giriza La Palma	IR	290	1,892,207	5881	0.0422	0.1103

References

Achard, F., Hugh, D.E., Stibig, H.J., Mayaux, P., Gallego, J., Richards, T., Malingreau, J.P., 2002. Determination of deforestation rates of the world's humid tropical forests. Science 297, 999–1002.

Arango, R., Sánchez, E. 2004. Pueblos indígenas de Colombia en el umbral del nuevo milenio: población, cultura y territorios: bases para el fortalecimiento social y económico de los pueblos indígenas. Departamento Nacional de Planeación – DNP, Bogotá.

Armenteras, D., Rudas, G., Rodríguez, N., Sua, S., Romero, M., 2006. Patterns and causes of deforestation in the Colombian Amazon. Ecological Indicators 6, 353–368.

Barros Ferraz, S.F.De., Vettorazz, C.A., Theobald, D.M., Ramos Ballester, M.V., 2005. Landscape dynamics of Amazonian deforestation between 1984 and 2002 in central Rondonia, Brazil: assessment and future scenarios. Forest Ecology and Management 204, 67–83.

Berry, P., Huber, O., Holst, B., 1995. Floristic analysis and phytogeography. In: Steyermark, J., Berry, P., Holst, B. (Eds.), Flora of the Venezuelan Guayana, The Missouri Botanical Garden, vol. 1. Timber Press, Portland, pp. 161–191.

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

Butler, R.A., Laurance, W.F., 2008. New strategies for conserving tropical forests. Trends in Ecology and Evolution 23 (9), 469–472.

Cardille, J.A., Foley, J.A., 2003. Agricultural land-use change in Brazilian Amazonia between 1980 and 1995: evidence from integrated satellite and census data. Remote Sensing of Environment 87, 551–562.

Chomitz, K.M., Thomas, T.D., 2003. Determinants of land use in Amazônia: a fine-scale spatial analysis. American Journal of Agricultural Economics 85, 1016–1028.

Davalos, L.M., 2001. The San Lucas mountain range in Colombia: how much conservation is owed to the violence? Biodiversity and Conservation. 10, 69–78.

DeFries, R., Hansen, A., Newton, A.C., Hansen, M.C., 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. Ecological Applications 15, 19–26.

Du Toit, J., Walker, B., Campbell, B., 2004. Conserving tropical nature: current challenges for ecologists. Trends in Ecology and Evolution 19, 12–17.

Etter, A., 1998. Mapa general de ecosistemas de Colombia (1:1.500.000). In: Chaves, M.E., Arango, N. (Eds.), Informe nacional sobre el estado de la biodiversidad 1997. Instituto de Investigación de Recursos Biológicos Alexander von Humboldt, PNUMA and Ministerio de Medio Ambiente. vol. 3, Bogotá, Colombia.

Etter, A., McAlpine, C., Pullar, D., Possingham, H., 2005. Modeling the age of tropical moist forest fragments in heavily-cleared lowland landscapes of Colombia. Forest Ecology and Management 208, 249–260.

Etter, A., McAlpine, C., Wilson, K., Phinn, H., 2006. Regional patterns of agricultural land use and deforestation in Colombia. Agriculture, Ecosystems, Environment 114, 369–386.

Fearnside, P.M., 1993. Deforestation in Brazilian Amazonia: the effect of population and land tenure. Ambio 22, 537–545.

Fearnside, P.M., 2005. Deforestation in Brazilian Amazonia: history, rates, and consequences. Conservation Biology 19, 680–688.

Foster, D., Fluet, M., Boose, E., 1999. Human or natural disturbance landscape -scale dynamics of tropical forests of Puerto Rico. Ecological Applications 9, 555–572.

Gascon, C., Williamson, B., Fonseca, G.A.B., 2000. Receding forest edges and vanishing reserves. Science 288, 1356–1358.

Guyana Shield Initiative (GSI), 2002. Conservation Priorities for the Guayana Shield. Priority Setting Workshop, Paramaribo, Suriname.

Hockings, M., Stolton, S., Dudley, N., 2000. Evaluating Effectiveness: A Framework for Assessing the Management of Protected Areas. Best Practice Protected Area Guidelines Series No. 6. IUCN, Gland, Switzerland and Cambridge, UK.

Huber, O., 1994. Recent advances in the fhytogeography of the Guayana Region, South America. Mémoires de la Société de Biogéographie 3(4), 53–63.

Joppa, L.N., Loarie, S.R., Pimm, S.L., 2008. On the protection of "protected areas". PNAS 105, 6673–6678.

Kirby, K.R., Laurance, W.F., Albernaz, A., Schroth, G., Fearnside, P.M., Bergen, S., Venticinque, E.M., da Costa, C., 2006. The future of deforestation in the Brazilian Amazon. Futures 38, 432–453.

Laurance, W.F., Albernaz, A.K.M., Da Costa, C., 2001. Is deforestation accelerating in the Brazilian Amazon? Environmental Conservation 28, 305–311.

Mas, J.F., 2005. Assessing protected areas effectiveness using surrounding (buffer) area environmentally similar to the target area. Environmental Monitoring and Assessment 105, 69–80.

Meidinger, D.V., 2003. Protocol for accuracy assessment of ecosystem maps, Res. Br. B.C. Min. For. Victoria, B.C. Tech. Rep. 011.

Myers, N., Mittermeier, R.A., Mittermeier, CG., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 852–858.

Naughton-Treves, L., Holland, M., Brandon, K., 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. Annual Review of Environment and Resources 30, 219–252.

Nepstad, D.C., Verýssimo, A., Alencar, A., Nobre, C., Lima, E., Lefebvre, P., Schlesinger, P., Potter, C., Moutinho, P., Mendoza, E., Cochrane, M., Brooks, V., 1999. Large-scale impoverishment of Amazon forests by logging and fire. Nature 398, 505–508.

Nepstad, D., Schwartzman, S., Bamberger, B., Santilli, M., Ray, D., Schlesinger, P., Lefebvre, P., Alencar, A., Prinz, E., Fiske, G., Rolla, A., 2006. Inhibition of Amazon deforestation and fire by parks and indigenous lands. Conservation Biology 20, 65–73.

Oliveira, P.J.C., Asner, G.P., Knapp, D.E., Almeyda, A., Galván-Gildemeister, R., Keene, S., Raybin, R.F., Smith, R.C., 2007. Land-use Allocation Protects the Peruvian Science 317, 1233.

Ponce, E., 1999. Áreas colectivas y territorios protegidos de comunidades indígenas y negras. Instituto Alexander von Humboldt, Bogotá D.C., Colombia.

Rodrigues, A.S.L., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D.C., da Fonseca, G.A.B., Gaston, K.J., Hoffmann, M., Long, J.S., Marquet, P.A., Pilgrim, J.D., Pressey, R.L., Schipper, J., Sechrest, W., Stuart, S.N., Underhill, L.G., Waller, R.W., Watts, M.E.J., Yan, X., 2004. Effectiveness of the global protected area network in representing species diversity. Nature 428, 640–643.

Rodríguez, N., Romero, M, Rincon, A., Bernal, N.R. 2006. Deforestación y fragmentación de ecosistemas naturales en el Escudo Guayanes colombiano. Instituto de Investigación de Recursos biológicos Alexander von Humboldt e instituto Colombiano para el Desarrollo de la Ciencia y la Tecnología "Francisco José de Caldas" – Colciencias, Bogotá D.C., Colombia.

Román-Cuesta, R.M., Martínez-Vilalta, J., 2006. Effectiveness of protected areas in mitigating fire within their boundaries: case study of Chiapas, Mexico. Conservation Biology 20, 1074–1086.

Romero, M., Galindo, G., Otero, J., Armenteras, D., 2004. Ecosistemas de la cuenca del Orinoco colombiano. Instituto de Investigación Recursos biológicos Alexander von Humboldt, Bogotá D.C., Colombia.

Rudel, T., 2006. Shrinking tropical forests and human agents of change. Conservation Biology 20, 1604–1609.

Rudel, T., 2007. Changing agents of deforestation: from state-initiated to enterprise driven processes, 1970–2000. Land Use Policy 24, 35–41.

Sánchez-Azofeifa, 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.

Schwartzman, S., Zimmerman, B., 2005. Conservation alliances with indigenous peoples of the Amazon. Conservation Biology 19, 721–727.

Schwartzman, S., Moreira, A., Nepstad, D., 2000. Rethinking tropical forest conservation: perils in parks. Conservation Biology 14, 1351–1357.

Sisben, 2003–2004. Información de la encuesta Sisben elaborada por el Departamento Nacional de Planeación años, 2003–2004.

Soares-Filho, B.S., Nepstad, D.C., Curran, L.M., Cerqueira, G.C., Garcia, R.A., Ramos, C.A., Voll, E., McDonald, A., Lefebvre, P., Schlesinger, P., 2006. Modelling conservation in the Amazon basin. Nature 440, 520–523.

Ter Steege, H., Zagt, R., Bertilsson, P., Singh, J., 2000. Plant diversity in Guyana: implications for the establishment of a Protected Areas system. In: ter Steege, H. (Ed.), Plant Diversity in Guyana. With Recommendation for a Protected Areas Strategy. Tropenbos Series 18. Tropenbos Foundation, Wageningen, The Netherlands. pp. 159–178.

United Nations Office on Drugs and Crime (UNODC), 2006. Colombia: Monitoreo de Cultivos de Coca. pp. 116.

Viña, A., Echavarria, F., Rundquist, D.C., 2004. Satellite change detection analysis of deforestation rates and patterns along the Colombia–Ecuador border. Ambio 33, 118–125.

Walter, H., 2002. Walter's Vegetation of the Earth: The Ecological Systems of the Geo-biosphere. Springer, Berlin, Germany.