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Characterizing a tropical deforestation wave: a dynamic spatial analysis of a deforestation hotspot in the Colombian Amazon

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Abstract

Tropical deforestation is the major contemporary threat to global biodiversity, because a diminishing extent of tropical forests supports the majority of the Earth's biodiversity. Forest clearing is often spatially concentrated in regions where human land use pressures, either planned or unplanned, increase the likelihood of deforestation. However, it is not a random process, but often moves in waves originating from settled areas. We investigate the spatial dynamics of land cover change in a tropical deforestation hotspot in the Colombian Amazon. We apply a forest cover zoning approach which permitted: calculation of colonization speed; comparative spatial analysis of patterns of deforestation and regeneration; analysis of spatial patterns of mature and recently regenerated forests; and the identification of local-level hotspots experiencing the fastest deforestation or regeneration. The colonization frontline moved at an average of 0.84 km yr^{-1} from 1989 to 2002, resulting in the clearing of 3400 ha yr^{-1} of forests beyond the 90% forest cover line. The dynamics of forest clearing varied across the colonization front according to the amount of forest in the landscape, but was spatially concentrated in well-defined 'local hotspots' of deforestation and forest regeneration. Behind the deforestation front, the transformed landscape mosaic is composed of cropping and grazing lands interspersed with mature forest fragments and patches of recently regenerated forests. We discuss the implications of the patterns of forest loss and fragmentation for biodiversity conservation within a framework of dynamic conservation planning.

Keywords: Amazon, Colombia, deforestation, forest cover zones, local hotspots, spatial pattern, tropical forests, unplanned colonization, zoning

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Introduction

Tropical deforestation is the major contemporary threat to global biodiversity, with a shrinking resource of tropical forests supporting most of the Earth's terrestrial biodiversity (Laurance, 1999; Green *et al*., 2004). Land cover change, including tropical deforestation, is a complex process occurring at multiple spatial and temporal scales and showing emergent properties resulting from the cumulative action of multiple agents (Lambin *et al*., 2003; Parker *et al*., 2003). Rapid deforestation often occurs in concentrated areas, where human population and resource use, either planned or unplanned, increases the likelihood of land clearing. Lepers *et al*. (2005) concludes that between 1980 and 2000, the tropics experienced more frequent forest cover loss than other regions, with the largest concentration of deforestation occurring in the Amazon Basin and Southeast Asia, and to a lesser extent in the borders of the Congo Basin. Most of the confirmed 'high certainty' hotspots of deforestation with sustained annual clearing rates greater than 2%, were located in the periphery of the Amazon Basin. The border of Colombia and Ecuador in

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the northwestern Amazon Basin appears in this synthesis as one of the high certainty rapid deforestation fronts (Lepers *et al*., 2005). Lepers *et al*. (2005) also notes, while land cover monitoring has improved substantially from the 1990s, datasets are still incomplete in coverage, preventing an accurate global accounting of land cover change.

Areas of rapid land cover change, such as 'deforestation hotspots' in tropical colonization fronts, are of critical importance to biodiversity conservation, especially when they overlap areas of high species endemism such as 'biodiversity hotspots' (Myers, 1988, 2003). So far, both hotspot concepts, deforestation and biodiversity hotspots, have been mainly used to inform priorities at the global level, and are a useful top-down approach for concentrating conservation actions. However, environmental variability (e.g. soil fertility, topography, accessibility) within broad hotspots such as the Eastern Colombian Amazon, and dynamic socioeconomic conditions, create spatially and temporally heterogeneous deforestation patterns, that should be considered when predicting future land cover dynamics and allocating conservation resources (Luck *et al*., 2004).

Conservation planning is increasingly emphasizing the need for a more dynamic and flexible approach that recognizes landscape change (Costello & Polasky, 2004; Meir *et al*., 2004; Pressey *et al*., 2004; Wilson *et al*., 2005). Areas surrounding colonization fronts in the tropics are, by definition, highly dynamic. To optimize conservation planning in the vicinity of colonization fronts, it is critical to understand the spatial and temporal dynamics of the deforestation process within and adjacent to the more dynamic 'local hotspot' areas. Within the broadly identified hotspots at the global (10^6 km^2) or regional (10^4 km^2) levels we, therefore, need to identify where highest and more persistent areas of deforestation are located, and what factors best explain the occurrence of these areas. Knowledge about the speed and the direction of advancing colonization fronts is an important factor in anticipating future threats to biodiversity. It is also important to understand how spatial characteristics of deforestation such as size and connectedness of forest clearings and regeneration patches vary across and adjacent to local hotspots, as they have potential important consequences for disturbance processes and biodiversity conservation. By identifying and characterizing local hotspots within the umbrella of global hotspots, precise information on vulnerability of deforestation will help inform the design of better targeted conservation plans and budgets (Margules & Pressey, 2000; Wilson *et al*., 2005).

The dynamics of tropical colonization is further complicated by the counterbalancing processes of deforestation and regeneration (e.g. Soares-Filho *et al*., 2001; Nagendra *et al*., 2003; Etter *et al*., 2005b). Such knowledge is especially needed in detailed local-level studies and extended to regional-level studies, so as to inform conservation planning, ecological restoration and as input for climate change analysis. Although it is still difficult to predict because of the inherent complexity of the process and the lack of adequate land cover change data, tropical deforestation is known to have an important impact on climate change through changes in carbon budgets (Fearnside & Laurance, 2004; Houghton, 2005), changes in land surface properties such as albedo, stomatal resistance and surface roughness (Pielke *et al*., 2002). Houghton *et al*. (2001) and DeFries *et al*. (2002) point to quantification of deforestation and regrowth rates associated with tropical land use as the major source of uncertainty in the calculation of contemporary global carbon budget.

The Colombian Amazon is an example of an unplanned deforestation scenario (Etter & Andrade, 1987; Viña *et al*., 2004; Etter *et al*., 2005b). Although historically there were some government attempts to direct the colonization processes (Legrand, 1988), especially from the 1970s, rapid deforestation remains virtually uncontrolled. Reasons for this are partly idiosyncratic, as most remote rainforests in Colombia have been regarded as 'territorios nacionales' (national territories) and considered as frontier areas of little strategic value to policymakers. Densely forested regions have only been considered valuable during periods when resources such as quinine and rubber were in demand and acquired political and economic value for the country (Domínguez & Gómez, 1990).

Tropical deforestation has been addressed by many researchers in areas such as the Amazon Basin (Soares-Filho *et al*., 2001; Laurance *et al*., 2002; Mertens *et al*., 2004; Viña *et al*., 2004), Central America (Southworth *et al*., 2002; Nagendra *et al*., 2003) and Africa (Mertens & Lambin, 1997; Imbernon, 1999; Serneels & Lambin, 2001). However, a detailed understanding of the dynamic spatial patterns of colonization fronts is lacking, particularly for unplanned deforestation. We addressed the question: does the spread of an unplanned deforestation front follow a systematic emergent spatial pattern? To address this question, we conducted a detailed analysis of the spatial dynamics of land cover change in an internationally recognized hotspot of unplanned tropical deforestation in the Colombian Amazon. We used forest cover change data derived from multitemporal Landsat Thematic Mapper (TM) and Enhanced TM (ETM) imagery to identify local hotspots of deforestation, and the dynamics of the regional deforestation – pattern within which the local hotspots are embedded. The analysis had four specific objectives:

(i) measure the speed at which the deforestation front is advancing (ii) determine the spatial pattern of the deforestation and regeneration within the forest cover zones across the entire colonization front; (iii) quantify the spatial distribution of mature and recently regenerated forests, and their comparative probability of being cleared, and (iv) identify local hotspots of deforestation and forest regeneration. All four objectives are important to global change studies and conservation planning.

Methods

Study area

The western Amazon Basin in Colombia is subject to increasing colonization pressure because of the proximity to more densely populated Andean region. In this region, some 3.6 million ha of forests have been impacted by clearing and fragmentation, in an area extending some 100 km in eastward direction from the Andes (Etter, 1998). Our study focuses on a section of this deforestation front in the western portion of the Caquetá Department (Fig. 1). The study area covers 23 000 km^2 with an average altitude of 250 m. The climate is humid tropical with an average annual rainfall between 3500 mm in the west to 2500 mm in the east, with one or two months with less than 100 mm. There are two main physiographic units: (i) the undulating to rolling plains of the interfluves (88% of study area); and, (ii) the alluvial plains along the main rivers (12% of study area), which include the floodplain and the alluvial terraces. With the exception of the more recent alluvial deposits along the rivers, most of the soils are well drained, very acidic with deep profiles (Malagón *et al*., 1993). Tropical forest communities vary with physiographic conditions. The forests of the interfluves have higher floristic diversity, with up to 145 tree species $>$10 cm diameter at breast height (DBH) per ha, while forests subject to human disturbance and fragmentation have fewer tree species and more palms (Carvajal *et al*., 1993). The alluvial forests are spatially very heterogeneous because of a combination of microtopography and inundation patterns, but are less diverse, with higher counts of palms and lianas. The study area includes subcatchments of the Caguán, Orteguaza and Caquetá rivers that are navigable for most of the year, and provide transport routes for colonization, along with an expanding road network.

The capital of the Caquetá Department, Florencia, has 80 000 inhabitants. In addition, the area includes 17 municipalities, with 14 towns under 5000 inhabitants. The rural population of the Department for the 1993

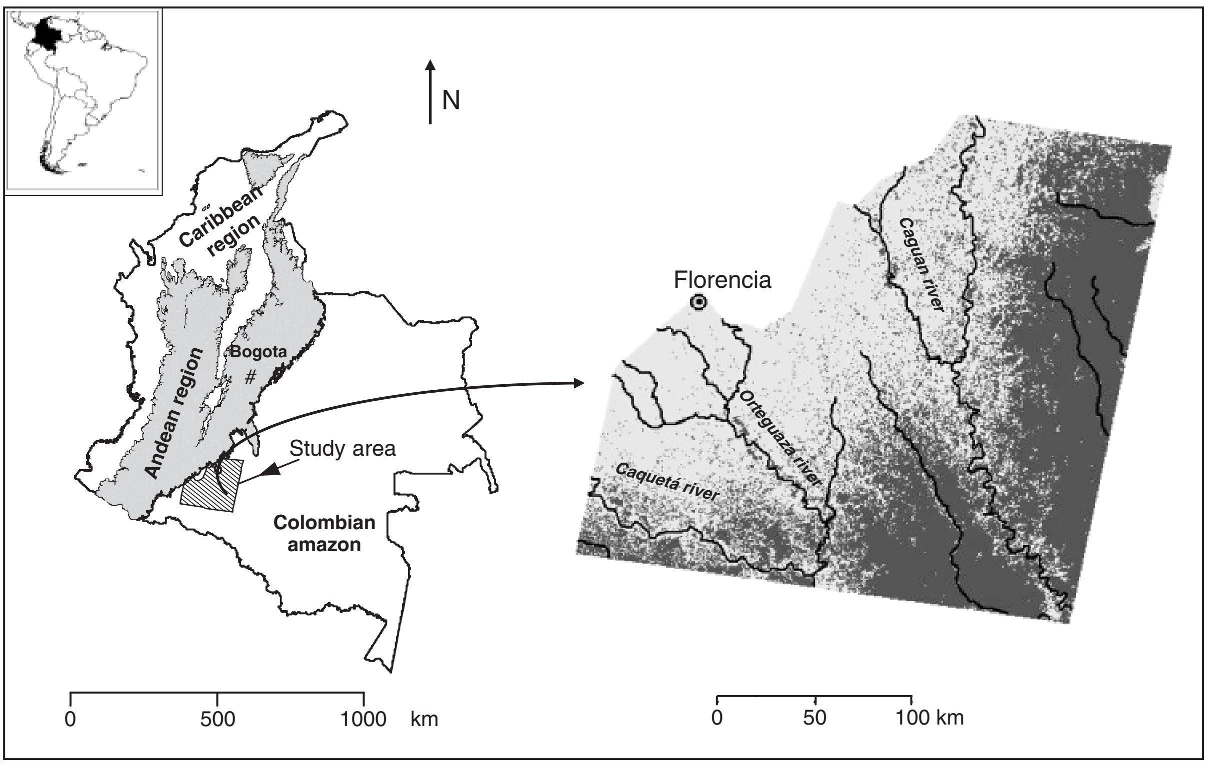

Fig. 1 Location of study area. The map of the study area shows the extent of the forest cover (dark gray) and the transformed areas (light gray) in 1989, the capital Florencia and the main rivers.

census was approximately 180 000, with an average density of 12 inhabitants/km^2 of settled land. The population of Caquetá has increased significantly over the last 40 years, because of the migration from the densely populated Andean region, in search of land and economic opportunities. In recent decades, Colombia has experienced considerable social and political unrest, driven by extremist left and right-winged armed forces. Frontier areas such as Caquetá have been the stronghold of guerrilla groups. This political instability has favored an upsurge in the number of illegal plantations of Coca (*Erythroxylum coca*), an important economic activity in frontier regions such as Caquetá during the last 20 years and a stimulant for migration of people to the region. The annual growth rate of the rural population in Caquetá between 1985 and 1993 reached 15% in some municipalities and was highly correlated with deforestation rates (Etter *et al*., 2005a). The latest national figures on illegal plantations, however, show an apparent, but debated drop of almost 50% from a high in 2001 (United Nations Office for Drug Control UNDOC, 2004). While Coca production is important for the cash economy, in terms of area, the most widespread land use in the Caquetá region is cattle grazing on pastures dominated by the African grass *Brachiaria* spp. In the western portion of Caquetá, closer to the towns and the capital Florencia, there is an expanding dairy farming industry, and the beginning of an oil palm plantation industry.

Land cover data

Four land cover maps were derived from the best available cloud-free Landsat TM and Landsat-7 ETM images, from 1989 (22 December), 1996 (11 August), 1999 (16 November) and 2002 (8 January), downloaded from the Global Land Cover Facility of the University of Maryland (http://glcfapp.umiacs.umd.edu:8080/esdi/index.jsp). All images were registered to the geo-referenced 2002 image using evenly spaced control points, to an average positional error (RMSE) of <0.3 pixels (Jensen, 1996). The procedures of geometric accuracy applied to the images are described in detail in Etter *et al*. (2005a). Using the ERDAS-Imagine 8.0 software, a supervised classification procedure (Jensen, 1996) was applied to differentiate four land cover classes: (a) Forests (well-drained and flood forests, including mature and recently regenerated forests); (b) Secondary vegetation (regenerated vegetation not yet forest); (c) Cleared land (comprising crops, pastures, fire, barren, urban); and (d) Water bodies (comprising rivers, oxbow and other lakes). The difference in acquisition season did not have an impact on the classification because the land cover classes where there could be potential spectral confusion (i.e. pastures, crops, burned areas and bare-soil) were grouped into one land cover class, 'cleared'.

The classification algorithm was trained on 20 representative sites of each land cover class selected using the senior author's knowledge of the area and the colonization process during the last 20 years (Etter & Andrade, 1987; Carvajal *et al*., 1993; Etter, 2001). This may lead to limitations in interpretation as not all land cover features can be accurately assigned to a class, but this was largely solved by using coarse well-defined classes (forest, secondary vegetation, cleared and water). Because of a lack of suitable higher spatial resolution image data and ground reference data, along with the inability to access field sites for the different image dates, validation could only be conducted using the training sites and the senior author's knowledge of ground condition at these sites. A 'pseudo' or training error assessment was conducted by classifying the training sites using a supervised classification algorithm. The accuracy figure was based on the number of pixels within each training site that were correctly assigned to a class (Jensen, 1996). Although this is a biased approach to error assessment, and does not replace traditional postclassification error assessment, it is commonly used as a self check when field and reference data are not available. The pseudo error analysis does indicate the suitability of training sites for identifying each land cover class of each image. The accuracy of the training sites used was summarized in contingency tables which showed overall accuracy levels of $>95\%$ for all classes in all time steps. For analysis all images were thereafter resampled to a common 1 ha grid, and binary (1,0) forest–nonforest maps were produced for each date.

Using the sequence of binary forest/nonforest maps, the forest areas were also divided into two classes: 'mature forests' (grid cells that appeared as $1 =$ forest in all dates) and 'regenerated forests' (grid cells that changed from 0 to 1 in any period since 1989 and were still $1 =$ forest in 2002). This procedure allows a precise identification of 'regenerated forests' from that period, but excludes 'older regenerated forests' (>20 years) that grew back before 1989, since colonization started more than 50 years ago. All colonization in the area was directional from west to east and most of the forest in 1989 was outside of the colonization front and not cleared before. Therefore, our underestimation of the spatial extent of pre-1980 'regenerated forests' should be very low or negligible compared with the mature forests.

Analysis

The analysis was conducted in five steps. The first step produced zoning maps of the region (one for each period) stratified by proportion of remnant forest using the binary forest/nonforest maps (1,0). In order to reduce fine-scale variability, the maps were smoothed by producing new maps where each grid cell contained the proportion of forest grid-cells in a window of varying size. Several window sizes (5×5, 10×10, 20×20, 50×50, 100×100 cells) were tested, with the 50×50 (25 km^2) window proving the most suitable for the analysis as it captured the spatial pattern of the advancing front while reducing the 'speckle' associated with image data. The window centroid was categorized into forest cover proportions on a scale of 0 to 1 according to the proportion of forest cover in the 25 km^2 neighborhood. These maps were subsequently classified into 10 classes of 10% forest cover intervals to produce a forest percentage cover zone map for each date, hereafter referred to as a forest cover zone. These zones were used to stratify the analysis of the spatial and temporal patterns of forest cover dynamics in the region.

Second, we devise a novel method for calculating the moving speed of the forest conversion front. To measure the speed at which the transformation was progressing, we defined two forest cover thresholds: the 90% forest cover boundary and the 60% forest cover boundary. The 90% forest cover boundary was used as a proxy for the 'colonization frontline', which indicates the commencement of clearing/perforation and the presence of scattered fields. The 60% forest cover boundary was used as a proxy for the 'critical fragmentation zone', because this is a critical threshold below which landscapes become fragmented (Gustafson & Parker, 1992; Fahrig, 2001). Because of the irregular shape of the advancing front, the average distance covered by each of both zones was obtained by measuring the distance between the 1989 and 2002 lines of both zones, sampled along 30 transects taken every 10 km along the 300 km front. The average annual speed was then calculated for the 13-year time span.

Third, to quantify the spatial patterns and rates of change, the binary forest/nonforest maps from the successive mapping dates (1989, 1996, 1999, 2002) were used to produce transition maps of forest (including regenerated forests) to nonforest (deforestation), and nonforest/forest (regeneration) for the 1989–1996, 1996–1999 and 1999–2002 periods. Because of the short time intervals between the land cover maps, a transition to forest was only defined as 'regenerated forest' when it came directly from secondary vegetation; thus, direct transitions from 'cleared land' to 'forest' were eliminated and treated as errors. Kimes *et al*. (1998) discussed the difficulty of accurately mapping secondary forests, especially in relation to age. In our study, this problem was addressed using high-quality multitemporal images that allowed to verify the persistence of secondary vegetation over several time periods. To calculate the amount of deforestation and forest regeneration in each forest cover zone, we cross-tabulated the deforestation (cleared area) and regeneration maps (regenerated area) with the forest cover zone maps for the beginning of each period (1989 for 1989–1996; 1996 for 1996–1999; 1999 for 1999–2002). The annual rates of change were then calculated for each forest cover zone according to Puyravaud (2003):

$$\text{Rate} = \left(\frac{1}{t_2 - t_1}\right) \ln\left(\frac{Ai_2}{Ai_1}\right),$$

where Ai_1 is the cover of class i at an initial time (t_1) and Ai_2 is the cover of class i at a later time (t_2). The proportion of regenerated forest in 1989–1996 that was re-cleared in 1999 and 2002 was also calculated, by overlaying the regeneration maps with the subsequent deforestation maps.

To understand how the spatial pattern of forest perforation and regeneration vary along the transformation gradient, we calculated the mean patch sizes and their proportion of connectedness within a 500 m neighborhood for the patches of 'deforested clearings' and the patches of 'regenerated forests' for each period (McGarigal *et al*., 2002). This assessed the spatial contiguity/dispersion of both deforestation and regeneration for each forest cover zone.

Fourth, to evaluate how the colonization process is altering the natural balance of mature vs. regenerated forests in the landscape, we used the forest map of 2002 classified into mature and regenerated forests to investigate their likelihood of clearing. The likelihood of clearing was derived from a map of deforestation predictions for the year 2002 as described in Etter *et al*. (2005a). The map was generated from a logistic regression model applied to the binary 1999–2002 deforestation transition map using available information on four explanatory variables: soil fertility, accessibility (cost-distance function from rivers and roads) and land cover neighborhood (amount of surrounding forest and nonforest secondary vegetation). The comparative deforestation likelihood of the 'mature' and 'recently regenerated' forests was obtained by cross-tabulation of the 2002 mature/regenerated forest map with the predicted deforestation probability map for 2000.

Finally, to analyze the spatial dynamics of deforestation and regeneration, we compared how the forest cover zones changed for each time period (e.g. cells making transition from the 70–80% forest cover zone to 60–70% forest cover zone in a given period of time). We performed map cross tabulations of the forest cover zone maps for the 1989–1996, 1996–1999, 1999–2002 time-steps. This allowed us to classify zone transitions in a spatially explicit way, and to identify the intensity of change (negative for deforestation, and positive for regeneration) depending on the number of zone jumps per period. We defined hotspots of change only as those areas with 'high speed' transitions defined as areas showing two or more zone-jumps (e.g. from 30–40% zone to 10–20% zone). Because the length of the 1989–1996 period was about twice that of 1996–1999 and 1999–2002, the number of jumps of the 1989–1996 period was standardized by dividing by two.

Results

Speed of advancing colonization front

The multitemporal forest zoning maps show the colonization front advancing eastwards into the Amazon lowlands (Fig. 2). The maps also show the influence of rivers in shaping the overall direction of the moving colonization front. The annual net deforestation rates showed a peak of 40 400 ha in 1996–1999, increasing from 18 600 ha in 1989–1996, and declining to 23 830 ha in 1999–2002. However, the average areas, spatial pattern and width of the zones of the front remained relatively constant for all periods. Only the 0–10% forest cover zone increased in area and width, while the 90–100% forest cover zone shrank as a result of the eastward movement of the deforestation front.

Between 1989 and 2002, the 90% forest cover boundary, a proxy for the 'colonization frontline', moved east an average distance of 11.2 km (Fig. 3), representing a rate of 0.84 km yr^{-1}. The rate varied between 0.5–3.2 km yr^{-1}, depending on the location along the front. An average of 25 400 ha of forests was cleared each year during the 1989–2002 period, of which 3 400 ha yr^{-1} was cleared in the perforation process beyond the 90% frontline. When the same procedure was applied to the 'critical fragmentation zone' (60% forest cover line), the average distance moved was 0.77 km yr^{-1} for the 1989–2002 period, varying from 0.32 to 2.9 km yr^{-1} depending on location. Below this threshold the total area increased by $\sim$ 340 000 ha for the study period with 15 000 ha of remnant forests becoming spatially disconnected in a matrix of cropping and grazing land use.

Spatial patterns of deforestation and forest regeneration

The rate of net forest change within the colonization front was the result of the combined effect of deforestation and forest regeneration, and varied with the forest cover zones (Fig. 4a). The highest rate of deforestation occurred in the 50–80% forest cover zones, with rates exceeding 4% yr^{-1} in the 1996–1999 period (Fig. 4b). The forest regeneration rate peaked in the 20–50% forest cover zones, with annual rates of up to 0.9% (Fig. 4c),

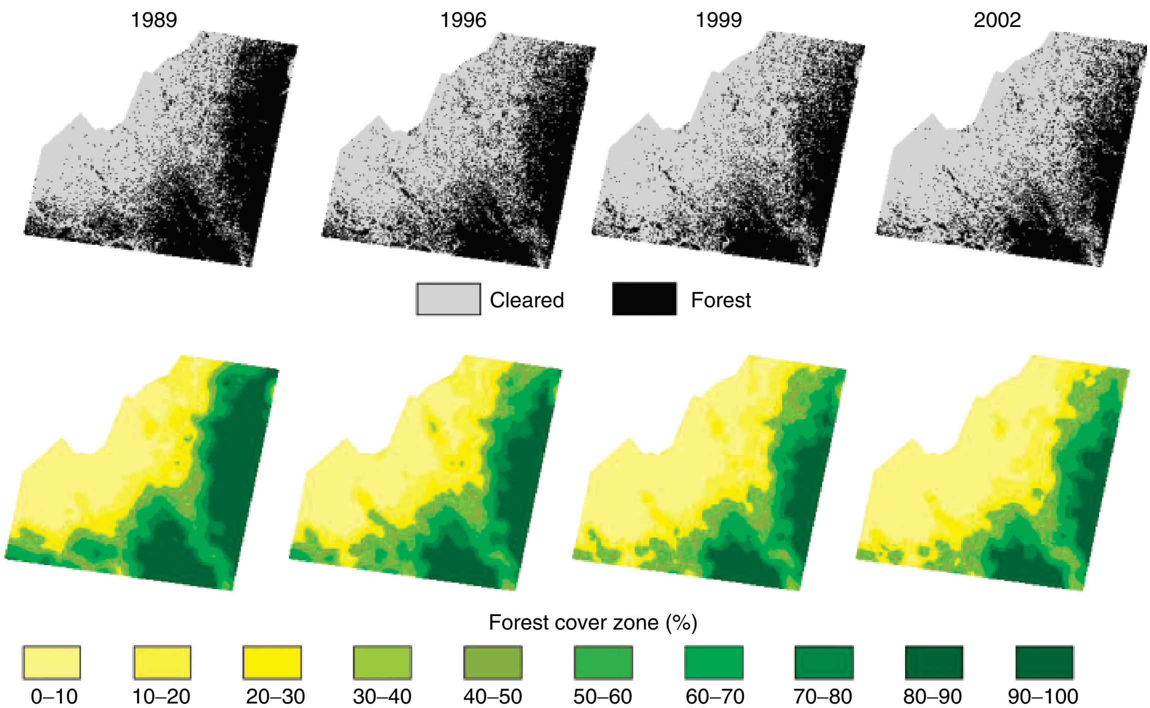

Fig. 2 Forest maps of the colonization front for each study date: (a) extent of forest cover (black = forest); and (b) percentage forest cover at 10% increment zones.

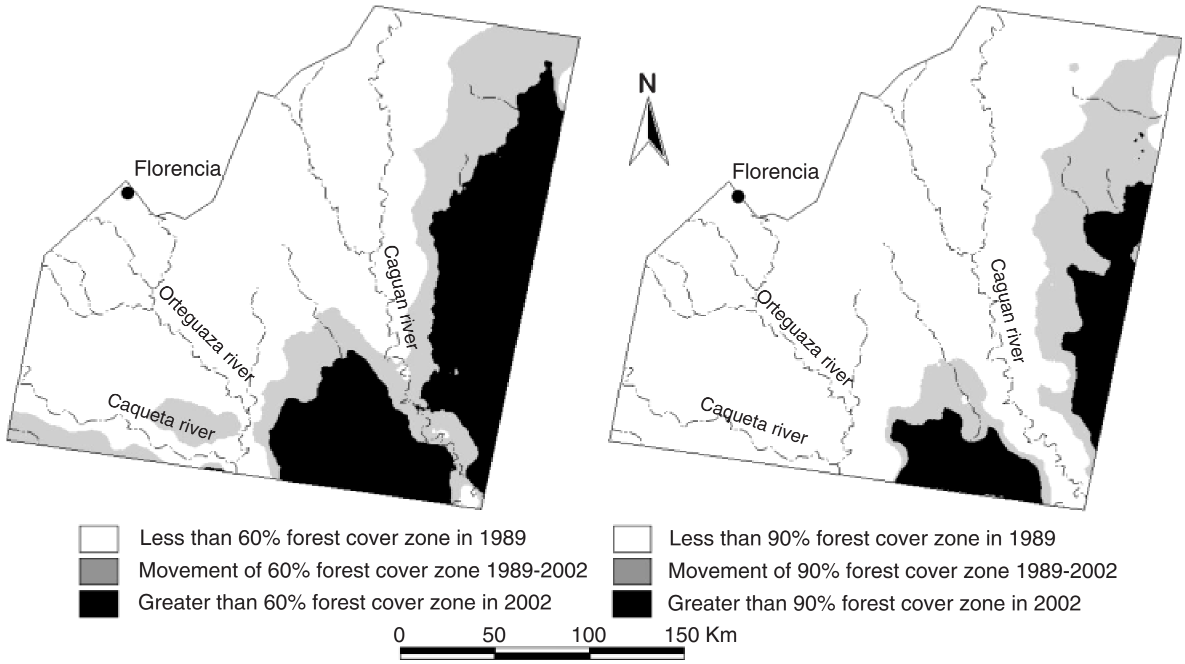

Fig. 3 Movement of the '90% forest cover line' (colonization frontline) and the '60% forest cover line' (critical fragmentation threshold) for the entire study period (1989–2002).

suggesting the progressive increase of secondary forests against 'mature forests' as the level of landscape transformation increases. The same pattern was maintained for all time periods (Fig. 4d).

The mean size of deforested areas (newly cleared fields) within the colonization front decreased from 4.5 ($\pm$ 0.47) ha in the 80–100% forest cover zones to 1.9 ($\pm$ 0.42) ha in the 0–20% forest cover zone (Fig. 5a). In contrast, the mean size of the newly regenerated forest patches was around 1.4 ($\pm$ 0.35) ha across all forest cover zones during the study period, which probably is the size of an average abandoned agricultural field. The connectivity of the newly deforested and regenerated patches exhibited a similar pattern across the forest cover zones, with a major loss of connectivity below the 20% forest cover line (Fig. 5b).

Comparing the regenerated forest maps (1989–1996, 1996–1999, 1999–2002) with subsequent deforestation maps (1996–1999, 1999–2002), showed that forests regenerating during the study period had a low stability. Of forests that regenerated during the 1989–1996, 62% were again cleared by 1999, and an additional 16.5% by 2002. Similarly, 45% of forests regenerating during the 1996–1999 period were cleared by 2002.

The logistic regression model for deforestation in 1999–2002 period applied to the 2002 forest cover as a proxy of the vulnerability to clearing, showed a higher vulnerability for regenerated than mature forests. The probability of clearing of recently regenerated forests (0.16 ± 0.023) was predicted to be twice as high as that of mature forests (0.07 ± 0.02). However, the regenerated forests make an increasing proportion of the forests in the more transformed landscapes. In the 2002 forest map the distribution forests showed a pattern of an increasing proportion regenerated forests with decreasing forest cover (Fig. 6).

Local hotspots of change

The analysis of the transitions of forest cover zones for the periods studied enabled us to produce a spatially explicit description of the dynamics of deforestation and regeneration (Fig. 7). A high rate of deforestation occurred in localized areas, or 'local hotspots' of forest loss and forest regeneration. A comparison of the three analysis periods indicates that some of the deforestation hotspots were spatially stable, such as the one in the northeast of the study area; others are advancing at a constant rate; while other locations show intense deforestation in one period followed by intense regeneration in a subsequent period, such as the south-western part of the study area. The more dynamic landscapes are invariably situated in the forest cover zones with an intermediate amount of remnant forests (30–70%). Between 1999 and 2002, there was an overall slowing of deforestation from a peak in 1996–1999. For that period, the most rapid deforestation occurred in the northeast of the study area, within the military exclusion zone of the failed peace process initiated in 1998 by the Pastrana government, an area where the FARC guerilla forces

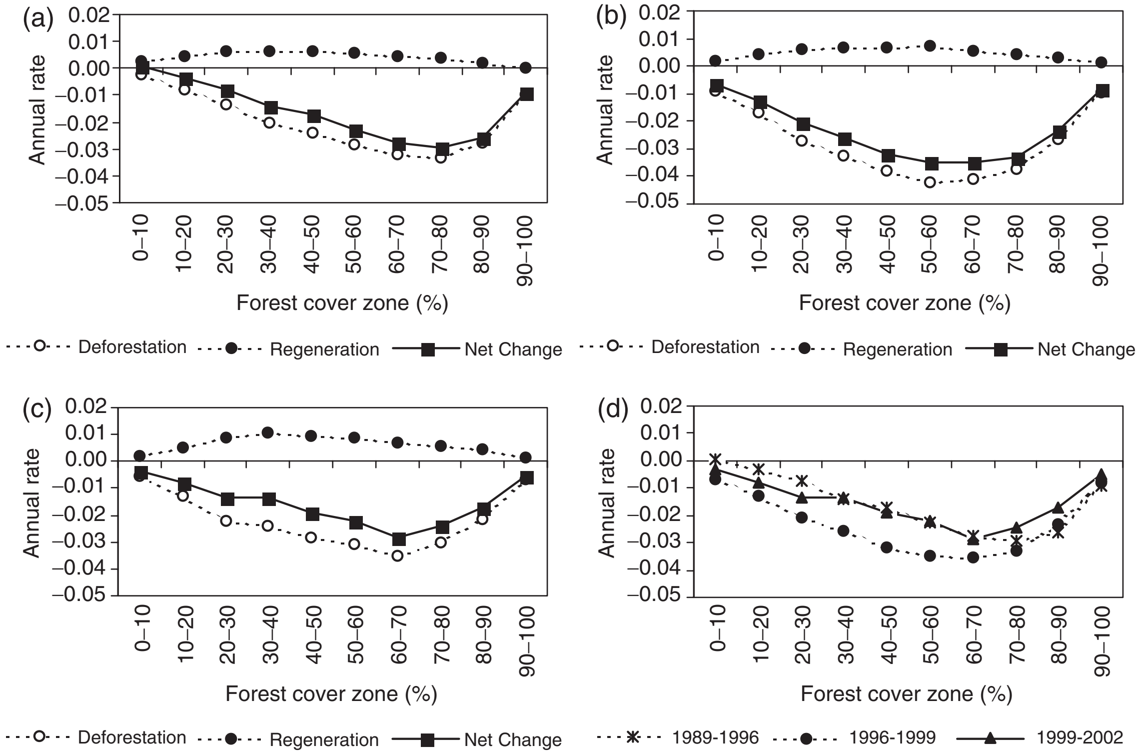

Fig. 4 Annual rates and spatial statistics of deforestation and forest regeneration by forest cover zones: (a) rates for 1989–1996; (b) rates for 1996–1999; (c) rates for 1999–2002; and (d) average net clearing rates by forest cover zone for the three periods.

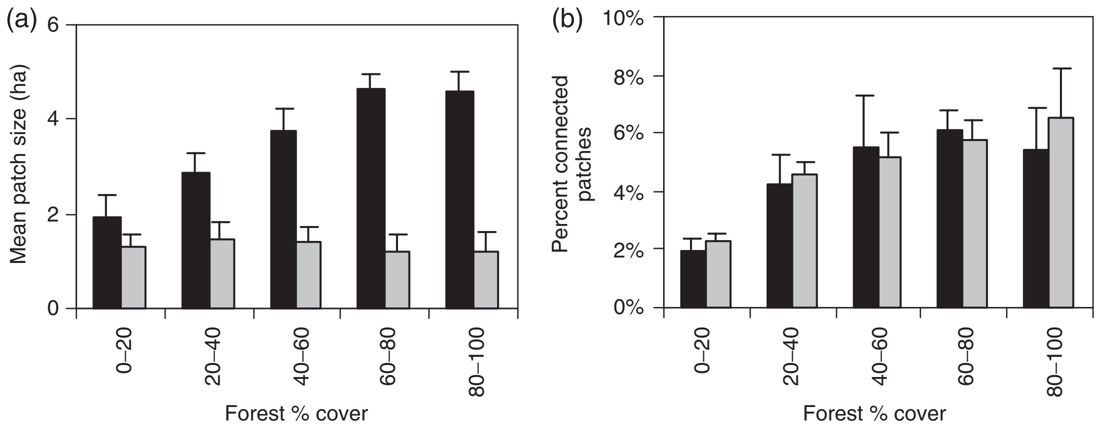

Fig. 5 Average values of selected landscape metrics: (a) Mean patch size; (b) Percent connected patches, of deforestation areas (black) and regeneration areas (gray). Error bars correspond to the standard deviation of the three periods.

were concentrated and controlled the countryside (Fig. 7). This area also coincided with the most rapid advance of the colonization front.

Discussion

Spatial and temporal patterns of colonization fronts

A complex and widespread process such as deforestation in the tropics needs to be studied at various spatial and temporal scales. Global studies provide broad brush estimates of deforestation by region but do not explain or explore medium and small scale spatial dynamics and causes (e.g. Achard *et al*., 2002; Lepers *et al*., 2005). Studies at the local-level, while too small to make generalizations, provide the necessary detail about the pattern and process of forest clearing relevant to conservation planning or ecological restoration. Regional-level studies, in turn, provide more precise knowledge about extent, rates and driving factors than

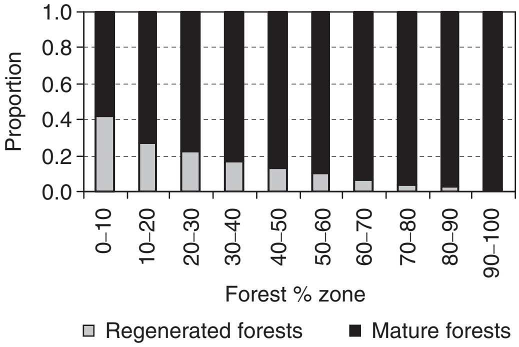

Fig. 6 Distribution of the proportion of mature and regenerated forests in the diiferent forest % cover zones for the year 2002.

global studies, and are the means to link to local studies (e.g. Lambin & Ehrlich, 1997; Laurance *et al*., 2002). It is at the latter two levels that our forest cover zoning method provides a novel approach to the analysis of the spatial and temporal dynamics of a tropical deforestation front. We provide a new approach for calculating the speed of deforestation across a colonization front in the tropics, which to our knowledge has not been done before. We consider this information as highly relevant to conservation and land use planning, for example, by planning the establishment of reserves well ahead of the arrival of the deforestation front. The movement of the deforestation front is the result of both new colonists moving into the area, and the periodic movement of existing colonists whose livelihood is made from clearing new land and selling the 'improvements' to land buyers (Etter & Andrade, 1987). Although the clearing of forests is the action of individual colonists, we have shown that their cumulative impact at a regional-level converges to produce an emerging property of a colonization front mimicking the movement of a wave.

The average annual rates of deforestation ranged from 2% to over 4% across the Caquetá region, broadly matching the results of previous studies (Sierra, 2000; Sanchez-Azofeifa *et al*., 2001; Steininger *et al*., 2001; Viña *et al*., 2004). We demonstrated however, that within this general movement pattern, the deforestation process at the local level occurred with varying intensities, with localized hotspots of rapid change, for both deforestation and forest regeneration. Also, that in 1999–2002, when peace talks with guerrillas took place in the Caquetá Department and part of it was demilitarized, important changes in patterns and rates occurred (Fig. 7). Overall, forest clearing slowed with respect to the previous 1996–1999 period (Fig. 4d). Rapid deforestation areas contracted, while the area of forest regeneration expanded substantially (Fig. 7). The former were concentrated in the military free zone of the northeastern part of the study area, while regeneration occurred around the military base of 'Tres Esquinas' north of the town of Solano. During this period, there was a substantial slowing of economic activity. These factors highlight how a political decision may have a direct and rapid influence on land cover dynamics.

The process of tropical forest transformation has recently been viewed as an interaction between deforestation and forest regeneration (Soares-Filho *et al*., 2001; Nagendra *et al*., 2003; Viña *et al*., 2004; Etter *et al*., 2005b). Our results go a step further by showing how these processes vary in space and time, and how they combine to form a successional forest mosaic in the transformed landscapes. Our approach to mapping forest transformation zones with defined rates of deforestation and regeneration is highly relevant to climate change studies, as it has the potential to improve the accuracy of current coarse resolution biomass accounting of impacted tropical forest regions (DeFries *et al*., 2002), where forest biomass is the primary source of terrestrial carbon emissions (Houghton, 2005).

The spatial patterns of deforestation and regeneration vary along the transformation gradient, with progressive proportion of secondary forests in the highly transformed landscapes (<40% forest cover) behind the colonization front. In a previous paper dealing with the characteristics of highly transformed landscapes of various lowland forest areas in Colombia (Etter *et al*., 2005b), we described the age structure of the remnant forest landscape at the local level, with recently regenerated forests constituting up to 80% of the forests in the high transformation stages. These observations, combined with the results of this study, indicate a highly fragmented distribution of both mature and secondary forests in a cropping and grazing land use matrix, with a general tendency towards a highly transformed and degraded tropical forest ecosystems, unable to support the full complement of native biodiversity that existed before the transformation process (Carvajal *et al*., 1993; Hill & Curran, 2001).

In the Colombian Amazon, the regional deforestation process is not random, but rather moves as a wave eastward into the Amazon Basin following the main rivers (Fig. 2). Our analysis of the forest cover zoning at the regional-level confirms, the relationships between the rates of deforestation, forest proportion in the landscape (Fig. 4d) and edge density described from local-level samples in previous research (Etter *et al*., 2005a). During deforestation, forest extent in any one area declines according to a logistic curve, with a slow initial deforestation phase followed by rapid loss and then a slow, semi-stable final phase at low levels of net forest cover. We advocate this as a general model for predicting the dynamics of unplanned tropical deforestation in

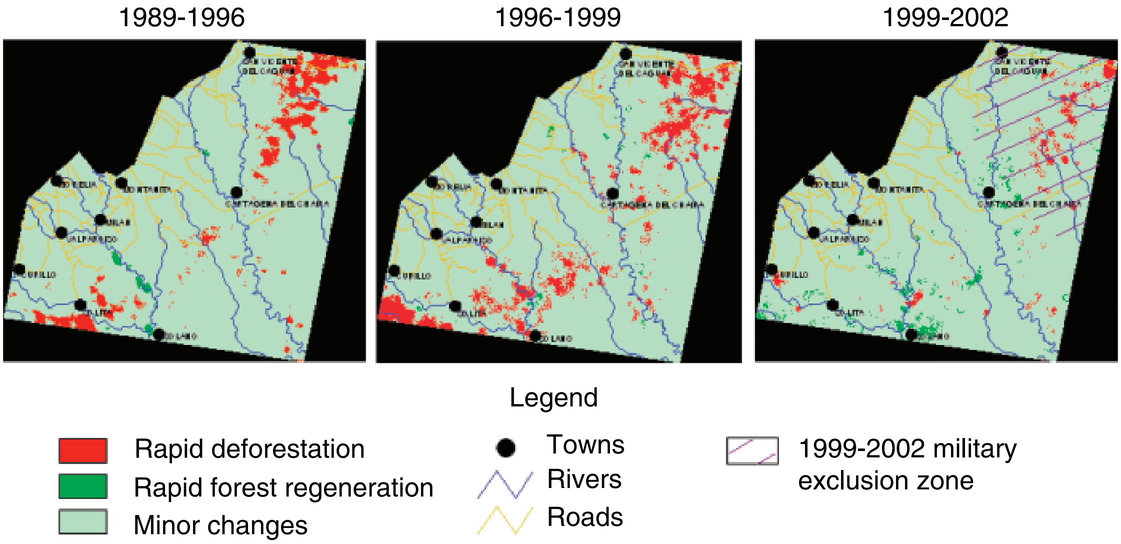

Fig. 7 Spatial location of the local hotspots of deforestation (red) and regeneration (green) for the three time periods of study.

the Colombian lowlands, but the applicability of this model requires further testing in other tropical regions (see Etter *et al.*, 2005a, for further details).

Although only 3400 ha yr^{-1} of the forest was cleared in the colonization frontline, it can be estimated that this initial forest perforation is exposing an additional 15 000 ha per year of adjacent forests to edge effects such as weed invasions, increased light penetration, plus further clearing pressures, timber extraction and hunting. We show that in Caquetá the end point of deforestation is a highly fragmented and biologically impoverished landscape with <10% of the original forests remaining and an average fragment size of <10 ha, with low connectivity. Carvajal *et al.* (1993) observed that fragmented forests in the Colombian Amazon have different floristic composition and reduced species richness. It is not known what proportion of the original biodiversity persists in transformed ecosystems. However, Hill & Curran (2001) found in tropical forests in Ghana, that forest area accounted for 92% of the variation in tree species richness for both mature and regenerating forests, indicating that fragment size is very important. We can infer from these studies that because of the very low forest proportion and small patch sizes of the highly transformed areas (<20% forest) in Caquetá, that the remnant forest and recently regenerated forest patches are unlikely to conserve the high floristic diversity of the original forests.

Limitations

Although the emerging macro-patterns of deforestation we describe are clear, this study does not attempt to explain how these patterns were generated by the socioeconomic and spatial interaction between local and regional agents. This needs to be done in the future in order to understand the underlying processes. Also, the availability of ground data could have improved the accuracy testing of the classified maps; nevertheless having large samples of 'mature forests' as spectral controls beyond the colonization front permitted an accurate spectral definition of 'forest', and the described patterns being mainly derived from the binary forest–nonforest map analysis, means that the observed spatial movement patterns of deforestation should be very solid. Finally, the research outcomes are specific to Colombia and may not be readily transferable to other unplanned tropical deforestation contexts (e.g. Bolivia, Cameroon, Peru); additional studies are needed to confirm the generality of these patterns.

Lessons for conservation planning

The same concept of concentrated 'hotspot' deforestation areas identified at a global (10^6 km^2) (Myers, 2003) or regional (10^4 km^2) (Lambin & Ehrlich, 1997) levels, can also be identified at the local-level (10^2 km^2) as we have shown. We believe that this local identification of hotspots may be more useful for more immediate, on-ground conservation action and monitoring.

The location of a site in the deforestation wave with respect to the timing of reserve selection has a big impact on the conservation outcome. According to Forman & Collinge (1997), spatial planning has the best conservation outcome when no more than 10–40% of the original natural vegetation has been removed, with conservation effectiveness decreasing rapidly below the 40% threshold. In the Caquetá colonization front, at least 30 000 ha of the mature, high biodiversity, tropical forests fall below this threshold annually. The area of 40–60% forest cover also experiences the highest rate of deforestation (Fig. 4). This means that there is little time for action once a forested area enters the deforestation wave. This knowledge can be used to design conservation strategies in advance of tropical colonization fronts. However, because colonization fronts inevitably lead to landscapes with a heterogeneous mosaic of mature and regenerated forests, such conservation strategies need be implemented in conjunction with ecological restoration and agricultural stewardship arrangements.

Conservation measures in a rapid deforestation scenario such as described here are difficult to design and implement. Probably the best way of saving viable tracts of forests in the region by legally declaring conservation reserves of at least 10–20 km in advance of the deforestation wave, which would leave adequate time to secure the borders in order to mitigate future threats of clearing by colonists. While reserves in this region may initially only be 'paper parks', Bruner *et al*. (2001) show in their worldwide study of the effectiveness of conservation areas, that legally established National Parks and Reserves are less likely to be cleared by colonists than the areas that are not protected. It is also important to educate and offer stewardship incentives to colonists to conserve a certain proportion of the mature forests (e.g. 40%) at the farm-level, and to integrate these measures into landscape-level conservation planning.

The theory of conservation planning is moving towards dealing with dynamic landscape change. This requires a sufficient understanding of the dynamics of the threatening processes in order to design, plan, and eventually implement, appropriate actions. As we develop tools for planning in a dynamic and uncertain world (Meir *et al*., 2004), we increasingly require spatially explicit information on threatening processes (Wilson *et al*., 2005). Projecting our model in space and time could generate spatially explicit threat maps. This can then be used to make wise conservation decisions that accommodate and anticipate rather than ignore threatening processes.

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