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Deforestation dynamics in a fragmented region of southern Amazonia: evaluation and future scenarios

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SUMMARY

The 'arc of deforestation' of southern Amazonia has one of the highest deforestation rates documented anywhere in the world. Landscape changes in a poorly studied but strategically important region in the Brazilian Amazon were studied using biennial Landsat TM/ETM+ images from 1984 to 2004. Deforestation rate for the period 1984–2004 was 2.47% yr^{-1} in the 7295 km^2 study area, but decreased to 1.99% and 2.15% in 2000–2002 and 2002–2004, respectively. Landscape structure changes were characterized by smaller forest patches that were further apart, but increasingly complex in shape. Deforestation was mainly driven by cattle ranching, which in turn was affected by distance to roads, with forest cover increasing at greater distances from roads. A multi-layer perceptron was used to develop future scenarios based on Markov Chain analysis. Based on current land use, forest cover in the region will decline from 42% in 2004 to 21% by 2016. Results indicate a critical threshold at 51% of forest cover in which landscape structure and connectivity changes abruptly. This suggests that the region requires greater efforts in environmental law enforcement, land-use planning and education programmes to maintain the remaining forest cover near this threshold.

Keywords: Alta Floresta, Amazon, deforestation, land-cover change, geographical information system, Mato Grosso

INTRODUCTION

Deforestation causing landscape change and loss of wildlife habitat is considered to be the most serious threat to global biodiversity (Sala *et al.* 2000). Deforestation has profound consequences for climate change (Meir *et al.* 2006; Gullison *et al.* 2007), biogeochemical cycles (Davidson & Artaxo 2004), and biodiversity in tropical, temperate and boreal regions (Gurd *et al.* 2001; Laurance *et al.* 2002*a*; Schmiegelow & Mönkköen 2002; Peres & Michalski 2006). Despite its importance, accurate estimates of deforestation rates are not available for most countries in the humid tropics (Grainger 1993), or the deforestation statistics from different sources are inconsistent (Hansen & DeFries 2004).

The Brazilian Amazon, which encompasses two-thirds of the Amazon basin, is the most extensive region of remaining tropical forest within a single country. However, annual deforestation rates have accelerated in recent years from 1.4 Mha in 1990 to 1.8 Mha in 1996, > 2.3 Mha in 2002 and > 2.7 Mha in 2004 (INPE [Instituto Nacional de Pesquisas Espaciais] 2008). This process continues to date with 0.7 Mha of forest cleared in August–December 2007 (INPE 2008). Since the 1970s, large-scale deforestation has been concentrated in the more accessible eastern, southern and south-western parts of the Amazon basin (Skole & Tucker 1993; Ferraz *et al.* 2005; INPE 2008) often generating a highly fragmented forest landscape containing forest remnants of varying size, shape, degree of connectivity and multiple disturbance regimes (Peres & Michalski 2006). Forest loss along this section of the Amazonian 'arc of deforestation' creates several types of landscape structure, ranging from the typical fish-bone pattern, in which small properties are regularly distributed along roads, to those dominated by sizeable remnants within extensive cattle ranches (Oliveira-Filho & Metzger 2006). Amazonian deforestation is likely to continue with further expansion of the cattle and soybean industries and other agricultural frontiers, so that 40% of the forest cover is likely to be converted by 2050 (Soares-Filho *et al.* 2006).

Different landscape patterns can influence the dynamics of populations, but the ecological consequences can differ depending on the pattern imposed on the landscape (Trani & Giles 1999). When a formerly continuous forest is isolated, the number of species will shift from its original equilibrium. This is affected by the area reduction in remaining forest patches and the distance to continuous forest or between patches (Laurance *et al.* 2002*a*). Declines in species diversity and abundance are usually related to the size of forest remnants and their degree of isolation (MacArthur & Wilson 1967; Diamond 1976; Simberloff 1976; Terborgh 1976). The larger the remaining forest area, the higher the original number of species remaining and the lower the rate of subsequent extinctions (Terborgh & Winter 1980).

Quantifiable changes in landscape structure, including land cover, remain an important aspect of landscape ecology because of their relationship with ecological processes (Turner 1989). Over the past decades, several metrics and indices have been developed to describe landscape configuration and composition (for example O'Neill *et al.* 1988; McGarigal &

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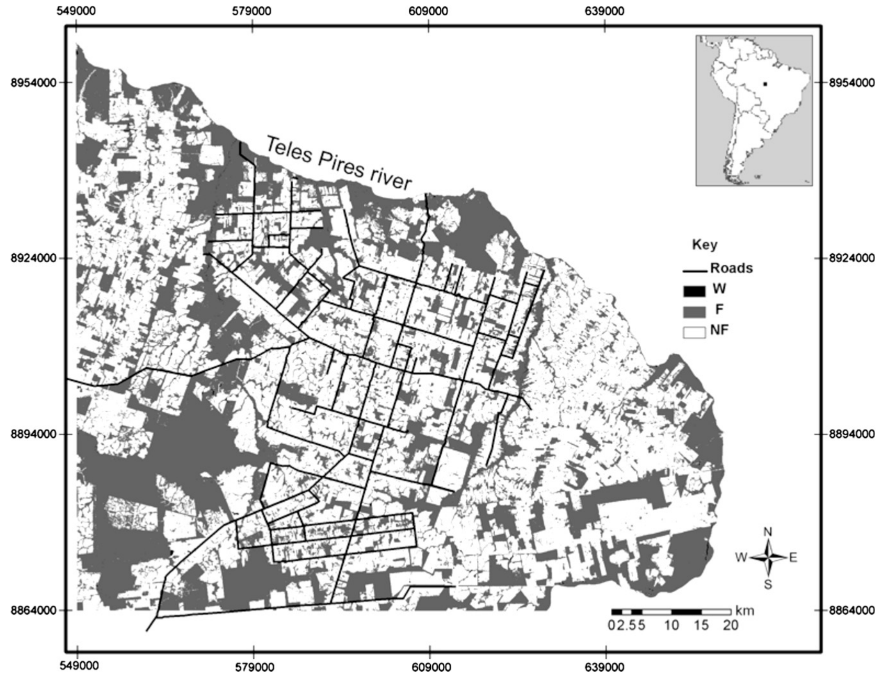


Figure 1 Location of the study region in Alta Floresta, northern Mato Grosso, Brazil, and the classified Landsat ETM image (227/67, 12 June 2004) showing the study area and the major paved and unpaved roads within the Alta Floresta municipal county. Grey and white areas on either bank of the Teles Pires River represent forest and non-forest cover, respectively.

Marks 1995; Gustafson 1998). These metrics can be used to characterize fragmentation, and include many quantifiable landscape changes describing a reduction in core habitat area, and an increase in edges, patch isolation and number of patches (Davidson 1998). Several studies in the Amazon have examined the impact of land-use changes upon landscape structure. However, few studies have considered temporal changes in landscape structure as deforestation progresses (Armenteras *et al.* 2006; Ferraz *et al.* 2005, 2006).

The main objective of this paper is to document levels of deforestation in a poorly studied region of the 'arc of deforestation' in the Brazilian Amazon and to examine which are the main socioeconomic drivers of deforestation in this area. In this paper we (1) document the impact of deforestation between 1984 and 2004 upon the landscape structure; (2) assess temporal changes in a number of landscape metrics related to remaining forest patches; (3) examine the relationships among local deforestation rates, the network of paved and unpaved roads, human population and head of cattle; and (4) develop possible future scenarios of land-use change for this region based on the rates and distribution of forest loss since 1984. Finally, we elaborate on the general implications of rapid agricultural frontier expansion and the long-term conservation implications in increasingly fragmented tropical forest landscapes.

METHODS

Study area

This study was conducted in the region of Alta Floresta, a prosperous frontier town located in northern Mato Grosso, southern Brazilian Amazonia (09° 53′ S, 56° 28′ W; Fig. 1). The study was of a 7295 km^2 area located to the south of the Teles Pires River, including the largest possible area in the region that could be considered free of natural *cerrado* scrublands, which could have been misclassified as deforested areas. These are located to the north and the south of the Landsat scene and are difficult to separate from anthropogenic non-forest areas using remote sensing. Areas to the north of the Teles Pires River were excluded because of major differences in land-use constrained by limited road access and a large protected area, both of which minimize deforestation rates. The study area exhibits three typical deforestation patterns, namely 'independent settlement', 'fishbone' and 'large property' (Oliveira-Filho & Metzger 2006).

This once entirely forested region has been subjected to one of the highest deforestation rates in the Brazilian Amazon since the early 1980s (INPE 2008; Peres & Michalski 2006), resulting in extensive areas of managed pastures and forest remnants. Deforestation in the region was primarily driven by agricultural colonization schemes and cattle ranching. The most recent estimates (based upon data for the 8947 km^2 Alta Floresta municipal county which largely overlaps our study area) are that the county contains 657 834 cattle (density of 74 cattle km^{-2}), and a human population of 47 190 (density of 5.3 inhabitants km^{-2}) (IBGE [Instituto Brasileiro de Geografia e Estatística] 2006, 2007).

Rates of deforestation

Landscape changes were analysed on the basis of a biennial sequence of 11 Landsat Thematic Mapper (1984–1998) and Enhanced Thematic Mapper Plus (2000–2004) images (scene 227/67) from 1984 to 2004 (21 June 1984, 11 June 1986, 18 July 1988, 06 June 1990, 11 June 1992, 03 July 1994, 22 June 1996, 28 June 1998, 27 July 2000, 30 May 2002, 12 June 2004) all with 30 m resolution. The images were registered and georectified to the 1996 satellite image to a positional error of < 10 m. The root mean square error of the georectification of the 1984–2004 images averaged 0.29 (SD = 0.03, range = 0.23–0.33) and we used an average of 14 points (SD = 3.8) to align the images. All data were projected on UTM 21S (datum SAD69). All images were classified as forest, non-forest and open water using band 1 (0.45–0.52 μm), band 2 (0.52–0.60 μm), and band 3 (0.63–0.69 μm). Following an unsupervised classification using the ISODATA clustering algorithm, we obtained 50 spectral classes, which were reclassified as forest, non-forest and open water based on our reference data points collected in the field and on our previous knowledge of the study area. A 3×3 mode filter was applied to all classified images to remove extraneous pixels that can result from random noise that are abnormally high or low relative to surrounding pixels. All analyses were undertaken in ArcGIS 9.1 (ESRI 2005). Land-use change was assumed to be an irreversible unidirectional loss of closed-canopy forests over time; once an area was recorded as deforested it could never be subsequently classified as forest. This approach was adopted because fallow land in the study region may return to a more natural state, but rarely if ever return to closed-canopy forests (F. Michalski & C. Peres, unpublished data 2004; see Sader & Joyce 1988). Furthermore, even 16 year-old secondary forests are structurally and spectrally difficult to distinguish from mature forest (Moran *et al.* 1996; Steininger 2000). Non-forest areas in this study are, therefore, defined as any area that had been subjected to clearcutting, including poorly managed or abandoned pastures that had subsequently regenerated to some extent.

We assessed the accuracy of the classified land cover map by comparing the 2004 reclassified image against 150 reference points collected in the field June–December 2004 in a fully representative full area map (for example core forest areas, pastures, forest edges and open areas within forest fragments). We eliminated some misclassifications (i.e. clouds) identified through visual interpretation with manual edition. Classified points and reference data points were compared to assess the accuracy of the land cover map using the Kappa Index of Agreement. Classifications also considered the most recent years coinciding with our ground-truth points, and compared the transitions between years.

Landscape structure

Landscape structure was characterized based on the classified Landsat images. We initially calculated 15 class and landscape-level structure metrics for each image using FRAGSTATS 3.3 (McGarigal & Marks 1995). Metric calculations were based on a 90 m (i.e. three pixels) edge depth, a 500 m search radius and connectivity threshold distance and an eight-neighbour rule (for patch delineation). The edge depth threshold was based on previous studies in Central Amazonia (Laurance *et al.* 2002*a*). The overall landscape boundary was not included as edge in calculations and no border was specified.

Class-metrics were selected based on previous studies assessing a large number of landscape structure metrics and indices (Coppedge *et al.* 2001; Neel *et al.* 2004). Many of the indices have been developed to capture the elements of patterns that are important to a specific ecological entity and can predict ecological processes exhibited at coarse spatial scales (Gustafson 1998). Moreover, habitat fragmentation is currently one of the most urgent challenges facing environmental planners (Carsjens & van Lier 2002). Following a preliminary analysis of the 15 metrics calculated, we selected mean patch area (AREA_MN), mean shape index (SHAPE_MN), mean proximity index (PROX_MN) and class cohesion index (COHESION) because of their importance in conservation and management (Gustafson 1998). AREA_MN describes the mean area of all forest patches (in hectares) comprising the landscape mosaic (McGarigal & Marks 1995). The number of original species in a forest area increases accordingly to the size of the remaining forest area (Terborgh & Winter 1980; Michalski & Peres 2005). SHAPE_MN, the simplest and most straightforward index of overall shape complexity, is defined as 1 when the forest patch is maximally compact (i.e., square or almost square) and increases without limit as patch shape becomes more irregular (McGarigal & Marks 1995). Shape of the forest fragments has a strong effect on the core area of the remaining fragments, which can affect the proportion of regenerating light-demanding pioneer trees as they have shown to increase in irregular fragments (Hill & Curran 2003). PROX_MN is a dimensionless index that describes the size and proximity of all forest patches whose edges are within a specific search radius of the focal patch, which is equal to zero if a patch has no neighbours within the specific search radius (Gustafson & Parker 1992). Even small clearings of < 100 m may be barriers for several rainforest species (Laurance *et al.* 2002*a*) and the proximity index clearly describes the structure of the landscape and the proximity of fragments of the same class. COHESION measures the physical connectedness of the corresponding patch type of any given focal class, and increases as forest patches become more clumped or aggregated in their distribution (Schumaker 1996; Gustafson 1998). Cohesion measures multiple aspects of the landscape (Gustafson 1998), is especially sensitive when the focal class has total area < 50% of the landscape (Neel *et al.* 2004) and is useful in predicting species dispersal in fragmented landscapes (Schumaker 1996).

Socioeconomic drivers of deforestation

Census data from 1990 to 2004 on the human population and the total bovine herd size (i.e. head of cattle) in the Alta Floresta municipal county (IBGE 2006, 2007) were used to investigate their relationships with the remaining area of forest cover.

In order to assess the impact of roads on deforestation, we examined the relationship between mean proportion of forest cover and distance from roads, which had been planned prior to the government-sponsored Alta Floresta settlement. For this analysis, digital maps of the major paved and unpaved roads available in different years of our time series were derived from 1:250 000 maps produced by the Alta Floresta Engineering Department (SOMAVI [Departamento de Engenharia e Obras, Prefeitura Municipal de Alta Floresta] 2000). Because these maps were available only for the municipal county of Alta Floresta, we restricted the analysis to a sub-sample of the study region that was located within its political borders (4024 km^2). In this analysis, we created a raster file of the straight-line distances to roads and cross-tabulated with the Landsat classified images of 2000, 2002 and 2004, corresponding to the years of the road maps. We then calculated mean forest cover ($\pm$ SD) using buffer intervals of 500 m from the nearest road, out to a distance of 22 km.

Predicting future scenarios

In order to predict possible future deforestation in the study region in the next 12 years, we used transition potential models that predict spatial distribution and not the total amount of deforestation using an extensively enhanced multi-layer perceptron (MLP) neural network followed by Markovian Chain analysis. A Markovian process is one in which the state of a system can be determined by knowing its previous state and the probability of transition from each state to each other state. Transition probabilities have been used extensively for analysis and modeling of land-use and land-cover change (Muller & Middleton 1994; Brown *et al.* 2000; Ferraz *et al.* 2005). We applied transition probabilities from forest to non-forest to predict the forest land-cover remaining in 2008, 2012 and 2016. We used the Kappa index agreement to validate our model using the 1984 and 1990 land-cover images as the earliest and latest images to predict the period of time when land-cover was known (1994 and 2004 land-cover). The Kappa index agreement measures the association between two input images. If the two input images are in perfect agreement (no changes has occurred), $K = 1$. In contrast, if the two images are completely different, $K = -1$ (Rosenfield & Fitzpatrick-Lins 1986). However, unlike the traditional Kappa statistic, this index breaks the validation down into several components, enabling specification of the location of change versus the amount of change. It is crucial to use Kappa to evaluate the effectiveness of the simulation because the per cent success due to chance alone can be substantial (Pontius *et al.* 2001).

The model was developed using a geographical information system (GIS) modelling programme (Clark Labs 1987–2006), and followed five sequential steps: (1) creation of land-cover change maps obtained by cross tabulating maps of forest cover from 1984 and 2004; (2) quantification of the relationships between deforestation and proximate causes and selection of the 'best' explanatory variables; (3) calibration of the model (training of the Artificial Neural Networks); (4) simulation (elaboration of a transition potential map of deforestation which predicts the spatial distribution of deforestation for the following period); and (5) assessment of the model performance by comparing actual and predicted deforestation.

The explanatory power test to select the 'best' variables was based on a contingency table analysis. The quantitative measure of association used was a Cramer's V (Ott *et al.* 1983), high values of which indicate a potentially strong explanatory value of the variable, with p values expressing the probability that the Cramer's V is not significantly different from zero. We integrated in the model two explanatory variables (maps) describing potential proximate causes of deforestation, namely distance from the nearest road (static variable) and distance from areas of existing disturbance in 1984 (dynamic variable). The latter map was created by extracting the disturbed areas from the earlier land-cover image, filtering it with a 3×3 mode filter to remove extraneous pixels and then running the same distance module used to calculate distance from roads. In order to predict the future deforestation map for the entire study region (7295 km^2) we used the 2004 Landsat image to confirm the road map from the Alta Floresta municipal area and extended the road network to neighbouring municipal counties (for which we did not have a road map) based on visible main roads. The same method has been used in previous studies to map unofficial roads in the Brazilian Amazon (Brandão & Souza 2006).

We calculated the total amount of deforestation in the modelled area based on the transition potential models using an extensively enhanced multi-layer perceptron (MLP) neural network followed by Markovian Chain analysis.

RESULTS

Land-use dynamic

Forest areas calculated for all 11 images indicate a rapid loss of forest during the study, as 6648 km^2 of the 7295 km^2 study area was still forested in 1984 (Fig. 2). Between 1984 and 2004, images showed 3607 km^2 of forest was lost, corresponding to a forest cover decline from 91.1% to 41.7% over 20 years. Our Kappa index of agreement for the 2004 map was 76.2%, being considered a substantial strength of agreement (Landis & Koch 1977).

Between 1984 and 2004, biennial deforestation rates in the Alta Floresta region averaged 4.94% and were almost linear ($r^2 = 0.997$, $p < 0.001$; Regression equation: $y = 0.0286x^4 - 227.38x^3 + 679080x^2 - 9E + 08x + 4E + 11$; Fig. 3). This is equivalent to a deforestation rate of 2.47% yr^{-1}. The proportion of non-forest land-cover (represented by managed and unmanaged pastures) had exceeded that of forest for the first time by 2000. The highest transition rates occurred between 1986–1988 (8.26%) and 1992–1994 (8.36%), whereas the slowest rates occurred between 2000–2002 (1.99%) and 2002–2004 (2.15%). This suggests a slight saturation in the post-2000 deforestation rates, when overall forest cover had already been reduced to 43.8% and when the Brazilian economy was under an import tariff policy reform process, which included several tariff increases in different sectors of the economy owing to the implementation of the Plano Real (Baumann *et al.* 1998).

Forest cover structure

Over the entire study period, mean forest patch area (AREA_MN) decreased from 85.3 ha (SD = 7468.7 ha,

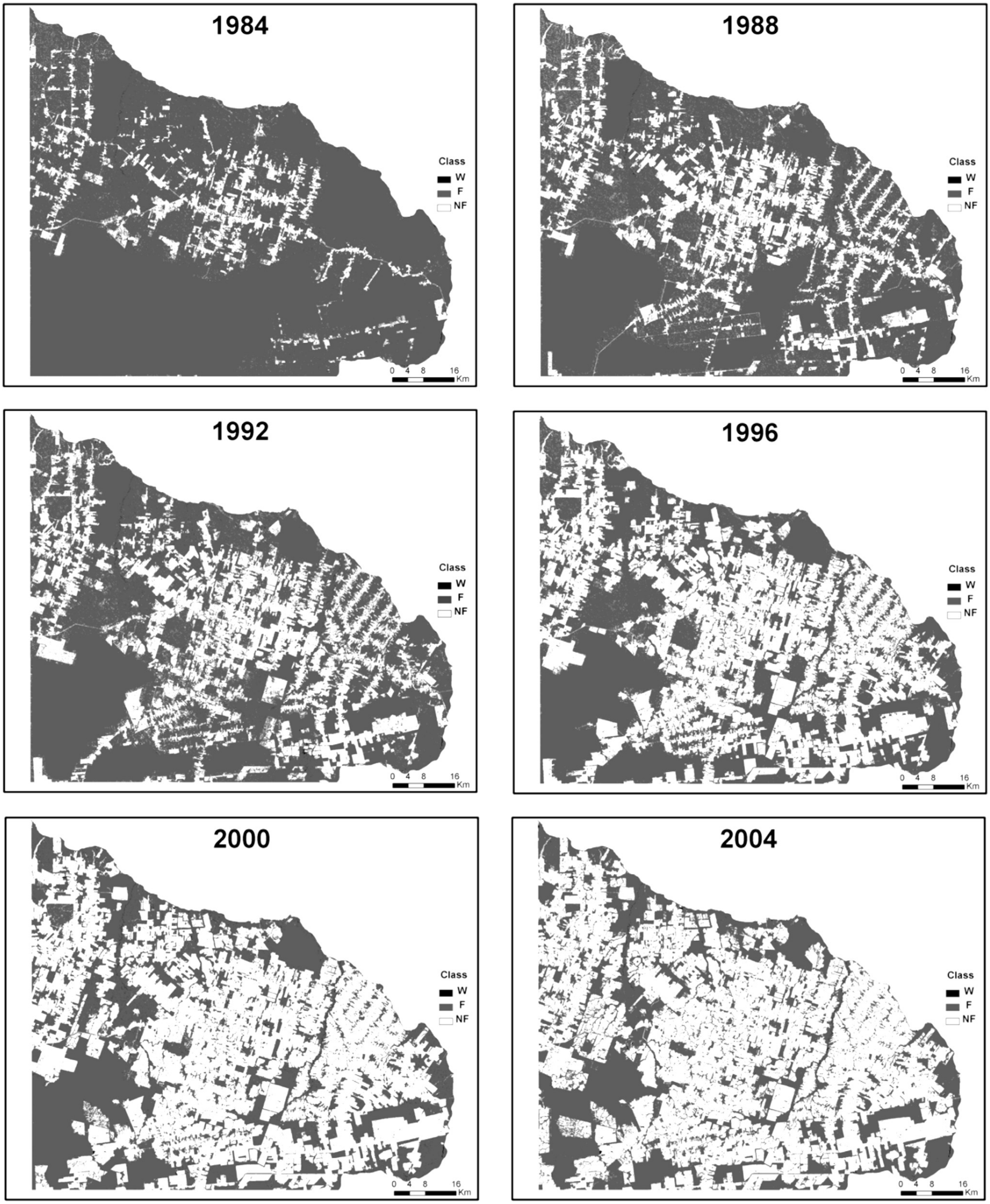


Figure 2 A series of land-use maps representing the evolution of the landscape structure in the Alta Floresta region at four-year intervals throughout our study period. Land-cover classes are represented by water (W), forest (F) and non-forest (NF).

$n = 7793$ patches) in 1984 to only 16.1 ha (SD = 643.0 ha, $n = 18932$ patches) in 2004 (Fig. 4*a*), whereas the mean shape index of forest patches (SHAPE_MN) increased as their perimeters became more irregular (Fig. 4*b*). The mean proximity index decreased over time as patch neighbourhoods (defined by the 500 m search radius) became more deforested,

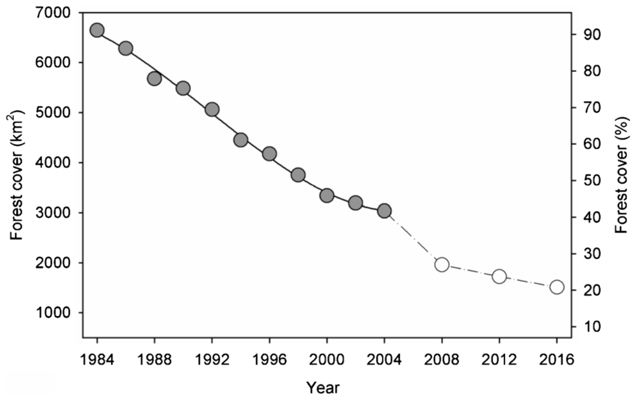


Figure 3 Forest cover (in km^2 left hand, % of original cover right hand) observed (grey circles) and predicted between 1984 and 2016 (open circles) in the study area.

with remaining inter-patch distances increasing as patches became less contiguous (or more fragmented) in their spatial distribution (Fig. 4*c*). Forest patch cohesion decreased gradually until 1998 but abruptly thereafter, when the proportion of the landscape comprising forest decreased suddenly and became increasingly subdivided (Fig. 4*d*).

Socioeconomic drivers of deforestation

There was a strong negative correlation between the total bovine cattle population and the remaining forest cover in the Alta Floresta municipal county for the period 1990–2004 ($r_s = -0.952, p = 0.000, n = 8$; Fig. 5). In contrast, the human

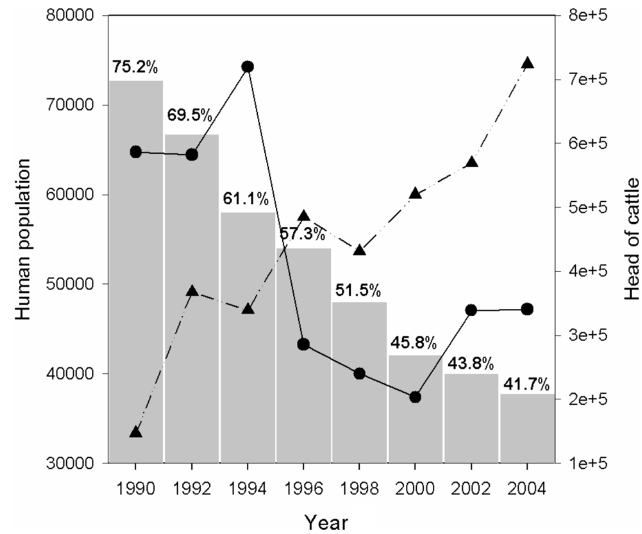
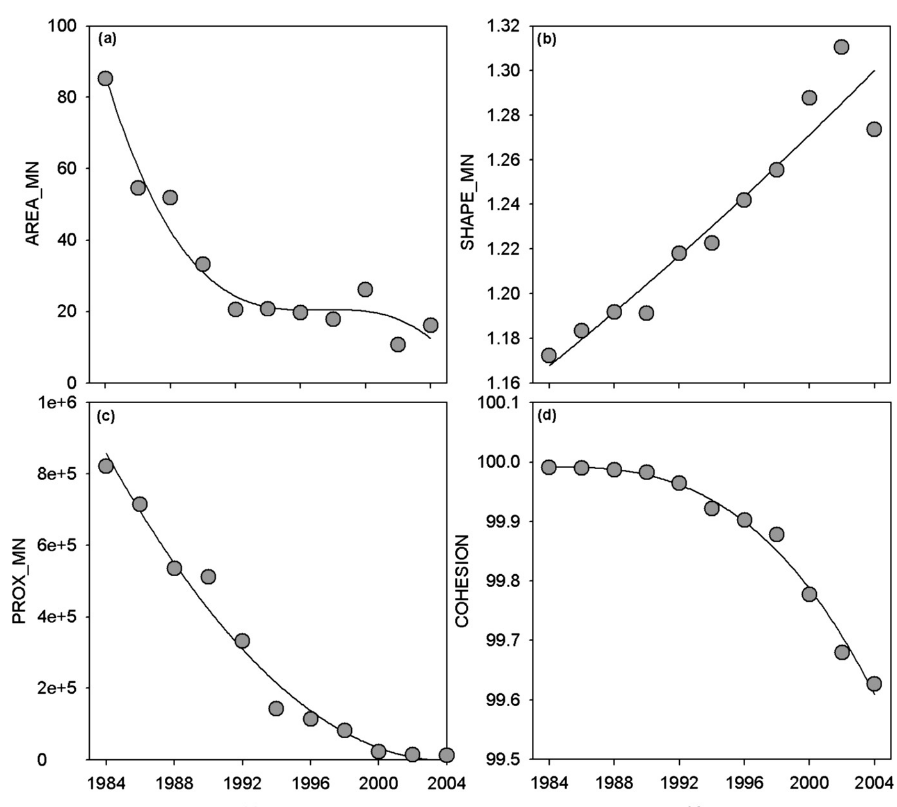


Figure 5 Forest cover area (grey bars, % of forest cover remaining above bars), human population (solid dots) and head of cattle (solid triangles) in Alta Floresta between 1990 and 2004.

population (both rural and urban) showed only a weak positive correlation with forest cover area for the same period ($r_s = 0.548, p = 0.160, n = 8$; Fig. 5). Not surprisingly, the Alta Floresta region attracted a marked influx of economic migrants from other Brazilian states when the region produced over 90 tonnes of gold from 1980 to 1996 (Hacon *et al.* 1995; Lacerda *et al.* 2004), but by the mid-1980s, the region still retained a large amount of forest cover (Fig. 3). Following a

Figure 4 Temporal variation (1984–2004) in class variables for forest land use: (*a*) mean patch area (AREA_MN); (*b*) mean shape index (SHAPE_MN); (*c*) mean proximity index (PROX_MN); and (*d*) patch cohesion index (COHESION).



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sudden decline in gold production owing to the depletion of easily mined deposits in 1994 (Hacon 1996), the human population decreased strikingly from 74 238 inhabitants in 1994 to 43 273 in 1996 (Fig. 5) as gold miners moved to new frontiers.

Considering the growing road network over the study period, there was a significant positive correlation between distance from the nearest road and the proportion of remaining forest cover for all of the three years for which an updated regional-scale road-map was available (2000: $r_s = 0.933$, $p = 0.000$, $n = 44$; Fig. 6*a*; 2002: $r_s = 0.880$, $p = 0.000$, $n = 44$; Fig. 6*b*; 2004: $r_s = 0.656$, $p = 0.000$, $n = 44$; Fig. 6*c*). Overall deforestation decreased with increasing distance from the road network; 71.3% and 62.1% of the deforestation in our study area occurred within 1.0 to 1.5 km of roads, respectively, a pattern similar to that observed in other Amazonia deforested regions (Alves *et al.* 1999; Ferraz *et al.* 2005). However, in recent years (2002 and 2004) there has been an extension of the unpaved road network and consequently less forest away from roads in general, indicating that the expansion of roads into more remote forest areas creates further access for logging companies and other sources of anthropogenic disturbances, and increases deforestation rates (Fig. 6*b*, *c*).

Future deforestation

Our model based on the current levels of deforestation predicted that non-forest cover in our study region by 2008, 2012 and 2016 will be 73%, 76% and 79% of total land-cover, respectively (Fig. 3). The models also showed that most of the deforestation will be concentrated within a 15–20 km radius from the town of Alta Floresta, where most of the existing disturbance in 1984 was already concentrated. The validation of our model, using 1984 and 1990 as the earliest and latest land-cover images to predict the 1994 and 2004 years, showed an accuracy rate of 62.92% and 52.85% after 5000 iterations and was able to correctly classify 49.08% and 41.23% of the grid cells for the entire map (Kappa = 0.79 and 0.66), respectively. For the specific transition from forest to non-forest we obtained an overall Kappa value of 0.13 and 0.11 for the 1994 and 2004 images, respectively.

We tested the explanatory power of our two driver variables of deforestation (non-forest land-cover) using 1984 and 2004 as the earliest and latest years. Distance from roads (Cramer's V = 0.23, $p < 0.001$) and distance from areas of existing disturbance in 1984 (Cramer's V = 0.23, $p < 0.001$) were useful variables for our model. The statistics showed that the accuracy rate of our model was 63.97% after 5000 iterations.

DISCUSSION

Deforestation rates and underlying causes

The recent history of forest cover change in an agricultural expansion frontier of the Amazonian 'arc of deforestation' shows that forests were lost at an alarming rate over a 20-yr period. Forest cover in this previously almost entirely forested region declined to only 42%, with a loss of 360 700 ha of undisturbed primary forest. The fastest clearcutting occurred in 1986–1988 and 1992–1994, with biennial forest losses exceeding 8%. Based on our predictions and on the current rates of change, only 21% of the original forest area

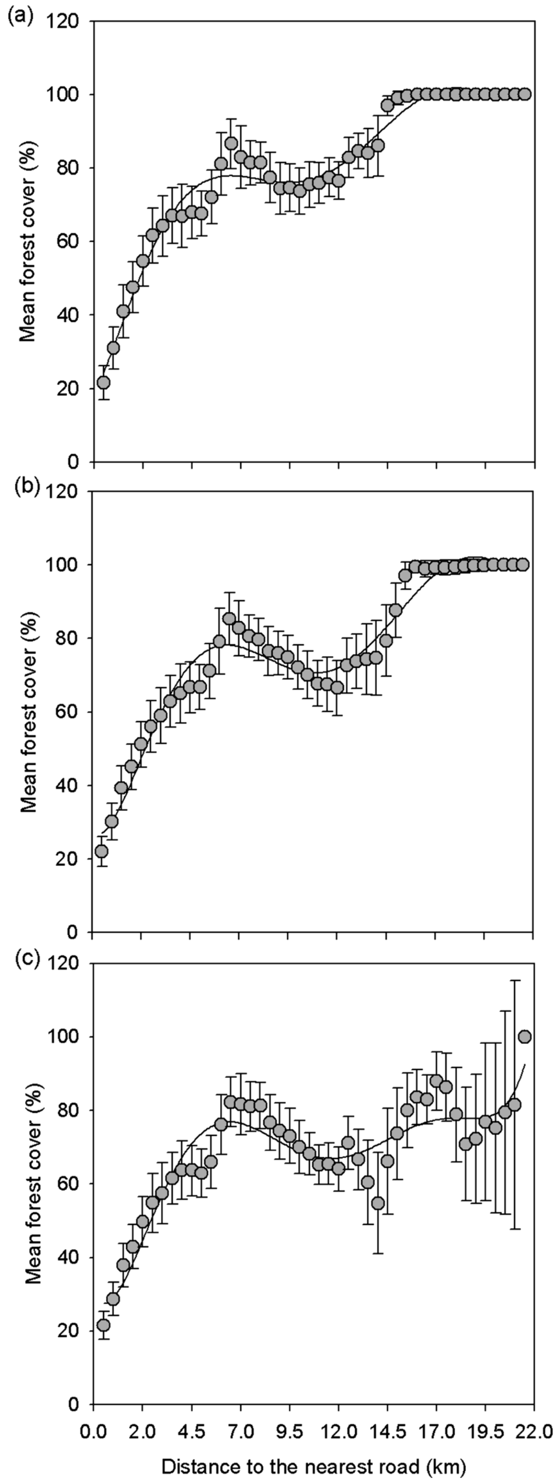

Figure 6 Relationship between mean (± SD) forest cover (%) and distance to the nearest paved or major unpaved road (km) in (*a*) 2000, (*b*) 2002 and (*c*) 2004 Landsat images. Updated road maps were only available for these three years.

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will remain in the region by 2016. The annual deforestation rate in the study area was clearly much higher than the average estimates of 0.4–0.5% reported for Latin America in the 1990s (Mayaux *et al.* 2005). This suggests that greater enforcement of existing environmental legislation should be directed to the Brazilian Amazon with especial emphasis to the 'arc of deforestation' region, where the highest deforestation rates continue to advance towards more central pristine areas of the Amazon basin. Moreover, rapid reporting from satellite monitoring programmes deployed by Brazilian government agencies and national and international non-governmental organizations should be ensured throughout the Amazon to tighten the enforcement of Brazilian forest legislation.

Land-cover change and structure in our study region is caused by different deforestation processes resulting in three distinct patterns (Oliveira-Filho & Metzger 2006). The first is the fishbone pattern, where small landholdings are regularly distributed along roads with deforestation typically originating from the road towards the back of the property. The second deforestation pattern occurs in small landholdings that are irregularly distributed across the wider landscape, rather than following roads. The third and most significant source of deforestation is driven by the creation of new pasturelands to service the burgeoning cattle ranching industry in large landholdings (Smeraldi & May 2008). Deforestation and forest fragmentation in the Amazon can be attributed primarily to cattle ranching, government-sponsored colonization projects, expansion of new highways and more recently to soybean farming (Soares-Filho *et al.* 2006), whereas forest degradation is primarily caused by industrial-scale logging (Asner *et al.* 2005). After cattle ranching, small-scale farmers account for the second most important driver of Amazonian deforestation (Kirby *et al.* 2006). Deforestation patches caused by both large and small landowners occurred throughout the study period, with small properties being located closer to the main towns whereas large properties containing larger forest patches were located further away. Forest fragmentation in large properties resulted from large clear-cuts, resulting in large forest patches with low connectivity (Oliveira-Filho & Metzger 2006). In terms of forest biodiversity conservation, the advantage of this landscape pattern is that large forest patches can retain a larger number of forest interior species with large area requirements (Michalski & Peres 2005; Lees & Peres 2006; Peres & Michalski 2006).

Changes in forest structure

The process of habitat fragmentation not only results in habitat loss, but also decreases patch sizes and increases the number of patches and the isolation between patches (Fahrig 2003). Our class metric results for forest structure revealed a typical fragmentation process with mean forest patch area decreasing during the study period, patches becoming increasingly subdivided and less physically connected. The same fragmentation pattern has also been observed in focal areas within our study area (Oliveira-Filho & Metzger 2006). Because of the fragmentation process, forest patches became more irregularly shaped throughout the study period, which was reflected in increasingly higher shape indices (McGarigal & Marks 1995). The proximity between forest patches also decreased as the neighbourhood became increasingly occupied by non-forest areas. Forest cohesion in our study area decreased rapidly after 1998, when the proportion of forest decreased abruptly and became increasingly subdivided. This observation can represent a critical tipping point of habitat fragmentation, where processes caused by forest loss started to affect the landscape dynamics and structure. By 1998, forest cover comprised only 51% of the landscape. This proportion could be considered as an approximate deforestation threshold for the study area and could be applied to other tropical forest regions following similar patterns of forest loss.

The process of decrease in mean forest patch area and subdivision of remaining forest habitat areas has been documented elsewhere, in North America (Coppedge *et al.* 2001) and in other parts of Amazonia (Ferraz *et al.* 2005). Following the same pattern observed for other fragmented landscapes, forest cohesion decreases rapidly and can predict spotted owl dispersal, and being therefore useful in distinguishing among different levels of landscape connectivity (Schumaker 1996). Similarly, this connectivity index can help in predicting dispersal of other species, contributing to maintenance of viable populations in fragmented landscapes.

Thresholds can be described as critical points after which changes occur rapidly. These critical thresholds of landscape structures vary according to different landscapes (Gardner *et al.* 1987) and abrupt changes in the structure of real landscapes occur over the course of the deforestation process at different proportions of habitat (Oliveira-Filho & Metzger 2006). For example, based on the fragmentation process in a watershed of the western Amazonian state of Rondônia, the threshold was around 35% of forest cover before the landscape structure changed abruptly (Ferraz *et al.* 2005), whereas a critical proportion of remaining habitat of 59.3% of the landscape was proposed by Stauffer (1985). Our suggested threshold of 51% of forest cover is slightly more conservative than the 60% suggested in a previous study for the same study region (Oliveira-Filho & Metzger 2006). This could be explained by the fact that we considered a much larger region (7295 km^2) than these authors (64 km^2) or because our findings are based on different connectivity metrics, which could be producing different outputs.

Socioeconomic drivers of deforestation

Our study reveals a strong correlation between the cumulative deforestation rates and the total number of cattle in Alta Floresta. Forest conversion into cattle pasture has been shown to be the primary driver of Amazonian deforestation in previous decades elsewhere (Hecht 1993) and more recently with improvements in beef production systems and changes in international markets (Nepstad *et al.* 2006). In our study area, we can safely assume that cattle ranching is the main driver of deforestation. However, the regional increase in the human population size during the gold mining peak of 1984–1992 (Hacon 1996) was followed by the highest forest conversion rate, which could be explained by the renewed influx of people and forest degradation associated with gold mining activities. After the boom-and-bust of gold production in 1994 there was a sudden decline in the human population, but bovine herds continued to increase throughout the region, maintaining high deforestation rates.

The opening and paving of roads driven by frontier colonization schemes result in a number of detrimental environmental effects in tropical forests (Brandão & Souza 2006; Kirby *et al.* 2006). Our study showed similar trends with higher deforestation rates occurring closer to existing roads. Overall deforestation in the Brazilian Amazon is primarily determined by human population density, highways and dry-season severity (Laurance *et al.* 2002*b*). However, the latest Landsat images revealed that forest clearance in Alta Floresta is not necessarily constrained by the main unpaved and paved roads, clearly showing an increase in deforestation in more remote areas. To some extent this is associated with private secondary roads created within large landholdings which provide access to remaining areas of primary forest located deep inside these properties. Many of these 'endogenous' roads are so well developed that they can be easily detected by Landsat images (Brandão & Souza 2006).

Deforestation and possible future scenarios

On the basis of the current land-use change, our model predicted that forest cover will be further reduced to only 21% of the study area in the next 12 years. The total amount of deforestation in the modelled area was based on the transition potential models using an extensively enhanced multi-layer perceptron (MLP) neural network followed by Markovian Chain analysis. The development of models of land-use change processes is an immense challenge for conservationists due to the complexity of interactions of environmental and socioeconomic effects on forest clearance (Mas *et al.* 2004). Our model was largely influenced by the original road network (static variable) and the distance from the main areas of disturbance in the earliest image (dynamic variable).

The evaluation of the performance of our model using the mapped propensity for deforestation for 1994 and 2004 forest compared with the actual deforestation observed during 1984–1994 and 1984–2004 showed that the model was able to correctly classify 49% and 41% of the pixels or an area corresponding to 3575 and 2991 km^2, respectively. The Kappa coefficient of the entire map was 0.79 and 0.66 but the same value for the transition forest to non-forest was much lower (0.13 and 0.11, respectively). Our Kappa coefficient was lower compared with other deforestation models (0.31–0.53 in Costa Rica; Pontius *et al.* 2001). This could be explained by the small number of variables used in our model. Other explanatory variables describing potential proximate causes of deforestation such as elevation, slope and soil types could improve the model considerably. However, we believe that overall, the results of our prediction are useful for future management plans of the study area and we recommend the creation of at least one protected area in the southern bank of the Teles Pires River to further reduce the availability of unprotected forests. On the basis of current deforestation rates the large tracts of undisturbed forest remaining in the south bank of this river are likely to shrink or disappear in the next decade. Moreover, the suggested threshold of 60% forest cover described in previous studies (Oliveira-Filho & Metzger 2006), or our more conservative estimate of 51%, had already been reached by 1998 in our study region and others parts of the Amazonian 'arc of deforestation'.

CONCLUSIONS

We showed that forest loss in a representative deforestation frontier of southern Amazonia has occurred at an alarming rate over the past 20 years. The Alta Floresta region is following a pattern of forest loss similar to that observed in other neotropical forest regions that were heavily fragmented much earlier (Laurance *et al.* 2002*b*; Ferraz *et al.* 2005; INPE 2008), even though this agricultural frontier is only 30 years-old, thus demonstrating how quickly deforestation thresholds can be reached and passed in tropical regions.

Landscape dynamics in our study area showed a systematic and relentless conversion of forest to non-forest areas (dominated by managed and unmanaged pasture). Landscape structure revealed a typical deforestation process resulting in habitat loss, decrease in patch sizes and increase in patch isolation (Fahrig 2003), and as expected deforestation rates decreased rapidly and non-linearly when moving away from main roads (Laurance *et al.* 2002*b*; Mas *et al.* 2004; Brandão & Souza 2006). As of 2004, the study area retained only 42% of forest cover and was already approaching a critical stage of deforestation. If observed deforestation probabilities remain unchanged, we predict that only 21% of the forest cover will remain in the study region by 2016, well below estimates of sustainable forest cover in the Amazon based on metrics of landscape structure (Ferraz *et al.* 2005; Oliveira-Filho & Metzger 2006), ecosystem services under future climate change scenarios (Cox *et al.* 2004; Avissar *et al.* 2006), and current levels of bird and mammal species persistence (Lees & Peres 2006; Peres & Michalski 2006; Michalski & Peres 2007). We suggest that the current proportion of forest cover should be maintained by ensuring the continuation of readily available data from satellite monitoring as well as *in situ* law enforcement of Brazilian forest legislation, especially during the dry season when most of the deforestation takes place.

If current rates of deforestation continue in this region, the region will reach a critical point where local extinctions of forest species will rapidly accelerate (Michalski & Peres 2005). Aggressive colonization frontiers can rapidly reach critical landscape structure thresholds beyond which more benign land-use alternatives such as the creation of new protected areas are no longer available unless significant efforts in capital-intensive habitat restoration can be deployed. A more cost-effective preventative approach should be followed based on greater enforcement to achieve wider compliance of private landholders with existing environmental laws. Additionally, the creation of protected areas in public land and environmental education of landowners in private areas can help maintain and conserve forests and biodiversity.

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References

Alves, D.S., Pereira, J.L.G., Souza, C.L., Soares, J.V. & Yamaguchi, F. (1999) Characterizing landscape changes in central Rondônia using Landsat TM imagery. *International Journal of Remote Sense* **20**: 2877–2882.

Armenteras, D., Rudas, G., Rodriguez, N., Sua, S. & Romero, N. (2006) Patterns and causes of deforestation in the Colombian Amazon. *Ecological Indicators* **6**: 353–368.

Asner, G.P., Knapp, D.E., Broadbent, E.N., Oliveira, P.J.C., Keller, M. & Silva, J.N. (2005) Selective logging in the Brazilian Amazon. *Science* **310**: 480–482.

Avissar, R., da Silva, R.R., & Werth, D. (2006) Impacts of tropical deforestation on regional and global hydroclimatology. In: *Emerging Threats to Tropical Forests*, ed. W. F. Laurance & C.A. Peres, pp. 67–79. Chicago, IL, USA: University of Chicago Press.

Baumann, R., Rivero, J., Zavattiero, Y. & ECLAC Brasilia office (1998) Tariffs and the Plano Real in Brazil. *Cepal Review* **66**: 171–185.

Brandão, A.O. & Souza, C.M. (2006) Mapping unofficial roads with Landsat images: a new tool to improve the monitoring of the Brazilian Amazon rainforest. *International Journal of Remote Sensing* **27**: 177–189.

Brown, D.G., Pijanowski, B.C. & Duh, J.D. (2000) Modeling the relationships between land use and land cover on private lands in the Upper Midwest, USA. *Journal of Environmental Management* **59**: 247–263.

Carsjens, G.J. & van Lier, H.N. (2002) Fragmentation and land-use planning – an introduction. *Landscape and Urban Planning* **58**: 79–82.

Clark Labs (1987–2006) Idrisi 15: The Andes Edition. The Idrisi Project. Clark University, Worcester, MA, USA [www document]. URL http://www.clarklabs.org

Coppedge, B.R., Engle, D.M., Fuhlendorf, S.D., Masters, R.E., & Gregory, M.S. (2001) Landscape cover type and pattern dynamics in fragmented southern Great Plains grasslands, USA. *Landscape Ecology* **16**: 677–690.

Cox, P.M., Betts, R.A., Collins, M., Harris, P.P., Huntingford, C. & Jones, C.D. (2004) Amazonian forest dieback under climate-carbon cycle projections for the 21st century. *Theoretical and Applied Climatology* **78**: 137–156.

Davidson, C. (1998) Issues in measuring landscape fragmentation. *Wildlife Society Bulletin* **26**: 32–37.

Davidson, E.A. & Artaxo, P. (2004) Globally significant changes in biological processes of the Amazon basin: results of the large-scale biosphere-atmosphere experiment. *Global Change Biology* **10**: 519–529.

Diamond, J.M. (1976) Island biogeography and conservation: strategy and limitations. *Science* **193**: 1027–1029.

ESRI (2005) ArcGis-version 9.1. Redlands, California, USA.

Fahrig, L. (2003) Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology Evolution and Systematics* **34**: 487–515.

Ferraz, S.F. de B., Vettorazzi, C.A., Theobald, D.M. & Ballester, M.V.R. (2005) Landscape dynamics of Amazonian deforestation between 1984 and 2002 in central Rondônia, Brazil: assessment and future scenarios. *Forest Ecology and Management* **204**: 67–83.

Ferraz, S.F. de B, Capao, L.M.S.A.C. & Vettorazzi, C.A. (2006) Temporal scale and spatial resolution effects on Amazon forest fragmentation assessment in Rondonia. *International Journal of Remote Sensing* **27**: 459–472.

Gardner, R.H., Milne, B.T., Turner, M.G. & O'Neill, R.V. (1987) Neutral models for the analysis of broad-scale landscape pattern. *Landscape Ecology* **1**: 19–28.

Grainger, A. (1993) Rates of deforestation in the humid tropics: estimates and measurements. *The Geographical Journal* **159**: 33–44.

Gullison, R.E., Frumhoff, P.C., Canadell, J.G., Field, C.B., Nepstad, D.C., Hayhoe, K., Avissar, R., Curran, L.M., Friedlingstein, P., Jones, C.D. & Nobre, C. (2007) Tropical forests and climate policy. *Science* **316**: 985–986.

Gurd, D.B., Nudds, T.D. & Rivard, D.H. (2001) Conservation of mammals in Eastern North American wildlife reserves: how small is too small? *Conservation Biology* **15**: 1355–1363.

Gustafson, E.J. (1998) Quantifying landscape spatial pattern: what is the state of the art? *Ecosystems* **1**: 143–156.

Gustafson, E.J. & Parker, G.R. (1992) Relationships between landcover proportion and indexes of landscape spatial pattern. *Landscape Ecology* **7**: 101–110.

Hacon, S. (1996) Avaliação do risco potencial para saúde humana da exposição de mercúrio na área urbana de Alta Floresta, MT, Bacia Amazônica, Brasil. PhD dissertation, Universidade Federal Fluminense, Niterói, RJ, Brazil: 172 pp.

Hacon, S., Artaxo, P., Gerab, F., Yamasoe, M.A., Campos, R.C., Conti, L.F. & Lacerda, L.D. (1995) Atmospheric mercury and trace elements in the region of Alta Floresta in the Amazon Basin. *Water, Air and Soil Pollution* **80**: 273–283.

Hansen, M.C. & DeFries, R.S. (2004) Detecting long-term global forest change using continuous fields of tree-cover maps from 8-km Advanced Very High Resolution Radiometer (AVHRR) data for the years 1982–99. *Ecossystems* **7**: 695–716.

Hecht, S.B. (1993) The logic of livestock and deforestation in Amazonia. *BioScience* **43**: 687–695.

Hill, J.L. & Curran, P.J. (2003) Area, shape and isolation of tropical forest fragments: effects on tree species diversity and implications for conservation. *Journal of Biogeography* **30**: 1391–1403.

IBGE (2006) Pesquisa Pecuária Municipal 1990–2005 [www document]. URL http://www.ibge.gov.br/home/estatistica/economia/ppm

IBGE (2007) Estimativas populacionais para os municípios brasileiros 1990–2006 [www document]. URL http://www.ibge.gov.br/home/estatistica/populacao

INPE (2008) Levantamento das áreas deflorestadas da Amazônia Legal no período 1988–2008: resultados. Ministério da Ciência e Tecnologia, São José dos Campos, SP, Brazil [www document]. URL http://www.obt.inpe.br/prodes/

Kirby, K.R., Laurance, W.F., Albernaz, A.K., 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.

Lacerda, L.D., de Souza, M. & Ribeiro, M.G. (2004) The effects of land use change on mercury distribution in soils of Alta Floresta, Southern Amazon. *Environmental Pollution* **129**: 247–255.

Landis, R.J. & Koch, G.G. (1977) The measurement of observer agreement for categorical data. *Biometrics* **33**: 159–174.

Laurance, W.F., Lovejoy, T.E., Vasconcelos, H.L., Bruna, E.M., Didham, R.K., Stouffer, P.C., Gascon, C., Bierregaard, R.O., Laurance, S.G. & Sampaio, E. (2002*a*) Ecossystem decay of Amazonian forest fragments: a 22-year investigation. *Conservation Biology* **16**: 605–618.

Laurance, W.F., Albernaz, A.K.M., Schroth, G., Fearnside, P.M., Bergen, S., Venticinque, E.M. & Costa, C. da. (2002*b*) Predictors of deforestation in the Brazilian Amazon. *Journal of Biogeography* **29**: 737–748.

Lees, A.C. & Peres, C.A. (2006) Rapid avifaunal collapse along the Amazonian deforestation frontier. *Biological Conservation* **133**: 198–211.

MacArthur, R.H. & Wilson, E.O. (1967) *The Theory of Island Biogeography*. Princeton, NJ, USA: Princeton University Press.

Mas, J.F., Puig, H., Palacio, J.L. & Sosa-López, A. (2004) Modelling deforestation using GIS and artificial neural networks. *Ecological Modelling and Software* **19**: 461–471.

Mayaux, P., Holmgren, P., Achard, F., Eva, H., Stibig, H. & Branthomme, A. (2005) Tropical forest cover change in 1990s and options for future monitoring. *Philosophical Transactions of the Royal Society B - Biological Sciences* **360**: 373–384.

McGarigal, K. & Marks, B.J. (1995) FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. USDA Forest Service GTR PNW-351, USA.

Meir, P., Cox, P. & Grace, J. (2006) The influence of terrestrial ecosystems on climate. *Trends in Ecology and Evolution* **21**: 254–260.

Michalski, F. & Peres, C.A. (2005) Anthropogenic determinants of primate and carnivore local extinctions in a fragmented forest landscape of southern Amazonia. *Biological Conservation* **124**: 383–396.

Michalski, F. & Peres, C.A. (2007) Disturbance-mediated mammal persistence and abundance-area relationships in Amazonian forest fragments. *Conservation Biology* **21**: 1626–1640.

Moran, E.F., Packer, A., Brondizio, E. & Tucker, J. (1996) Restoration of vegetation cover in the eastern Amazon. *Ecological Economics* **18**: 41–54.

Muller, M.R. & Middleton, J. (1994) A markov model of land-use change dynamics in the Niagara region, Ontario, Canada. *Landscape Ecology* **9**: 151–157.

Neel, M.C., McGarigal, K. & Cushman, S.A. (2004) Behavior of class-level landscape metrics across gradients of class aggregation and area. *Landscape Ecology* **19**: 435–455.

Nepstad, D.C., Stickler, C.M. & Almeida, O.T. (2006) Globalization of the Amazon soy and beef industries: opportunities for conservation. *Conservation Biology* **20**: 1595–1603.

O'Neill, R.V., Krummel, J.R., Gardner, R.H., Sugihara, G., Jackson, B., DeAngelis, D.L., Milne, B.T., Turner, M.G., Zygmunt, B., Christensen, S.W., Dale, V.H. & Graham, R.L. (1988) Indices of landscape pattern. *Landscape Ecology* **1**: 153–162.

Oliveira-Filho, F.J.B. & Metzger, J.P. (2006) Thresholds in landscape structure for three common deforestation patterns in the Brazilian Amazon. *Landscape Ecology* **21**: 1061–1073.

Ott, L., Larson, R.F. & Mendenhall, W. (1983) *Statistics: A Tool for the Social Sciences*. Boston, USA: Duxbury Press.

Peres, C.A. & Michalski, F. (2006) Synergistic effects of habitat disturbance and hunting in Amazonian forest fragments. In: *Emerging Threats to Tropical Forests*, ed. W.F. Laurance & C.A. Peres, pp. 105–127. Chicago, IL USA: University of Chicago Press.

Pontius, R.G., Cornell, J.D. & Hall, C.A.S. (2001) Modelling the spatial pattern of land-use change with GEOMOD2: application and validation in Costa Rica. *Agriculture, Ecossystems and Environment* **85**: 191–203.

Rosenfield, G.H. & Fitzpatrick-Lins, K. (1986) A coefficient of agreement as a measure of thematic classification accuracy. *Photogrammetric Engineering and Remote Sensing* **52**: 223–227.

Sader, S.A. & Joyce, A.T. (1988) Deforestation rates and trends in Costa Rica, 1940–1983. *Biotropica* **20**: 11–19.

Sala, O.E., Chapin, F.S.I., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. & Wall, D.H. (2000) Global biodiversity scenarios for the year 2100. *Science* **287**: 1770–1774.

Schmiegelow, F.K.A. & Mönkköen, M. (2002) Habitat loss and fragmentation in dynamic landscapes: avian perspectives from the boreal forest. *Ecological Applications* **12**: 375–389.

Schumaker, N.H. (1996) Using landscape indices to predict habitat connectivity. *Ecology* **77**: 1210–1225.

Simberloff, D. (1976) Experimental zoogeography: effects of island size. *Ecology* **57**: 629–642.

Skole, D. & Tucker, C.J. (1993) Tropical deforestation and habitat fragmentation in the Amazon: satellite data from 1978 to 1988. *Science* **260**: 1905–1910.

Smeraldi, R. & May, P. (2008) O reino do gado: uma nova fase na pecuarização da Amazônia. Amigos da Terra, Amazônia Brasileira, São Paulo, Brazil.

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.

SOMAVI (2000) Mapa do Município – Estradas vicinais e comunidades. 1:250 000. Alta Floresta, MT: Prefeitura Municipal.

Stauffer, D. (1985) *Introduction to Percolation Theory*. London, UK: Taylor and Francis.

Steininger, M.K. (2000) Satellite estimation of tropical secondary forest above-ground biomass: data from Brazil and Bolivia. *International Journal of Remote Sensing* **21**: 1139–1157.

Terborgh, J. (1976) Island biogeography and conservation: strategy and limitations. *Science* **193**: 1029–1030.

Terborgh, J. & Winter, B. (1980) Some causes of extinction. In: *Conservation Biology, and Evolutionary Ecological Perspective*, ed. M.E. Soulé & B.A. Wilcox, pp. 119–133. Sunderland, MA, USA: Sinauer Associates.

Trani, M.K. & Giles, R.H. (1999) An analysis of deforestation: metrics used to describe pattern change. *Forest Ecology and Management* **114**: 459–470.

Turner, M.G. (1989) Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics* **20**: 171–197.