Dynamics and patterns of deforestation in the western Amazon: the Napo deforestation front, 1986–1996

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Abstract

Although much emphasis has been given to the Brazilian Amazon in the analysis of tropical deforestation, information on other areas of the Amazon is scarce. This study examines the deforestation and level of forest fragmentation in one of the two deforestation fronts in the Amazon region outside Brazil: the Napo region of western Amazonia. Data were generated using satellite imagery for the years 1986 and 1996. Results show that while the extent and rate of deforestation in the region are important, they are less than previously reported. Forest fragmentation, however, adds a significant level of impact by affecting a very large proportion of the remaining forest cover. Better understanding of the dynamics and patterns of deforestation at regional, national and global levels requires the study of a representative number of areas. A regional focus, like that used here, facilitates this because most deforestation occurs in a small number of high-intensity deforestation fronts. © 2000 Elsevier Science Ltd. All rights reserved.

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Introduction

While tropical deforestation is widespread globally, the bulk of it seems to be taking place in a relatively small number of regions. According to Myers (1993)
there are 14 deforestation fronts in the world. Few, however, have regional databases providing current information on the dynamics and patterns of land cover change (Bierregaard, Lovejoy, Kapos, Dos Santos & Hutchings, 1992; Grainger, 1993; Meyers, 1994). Adequate geographic detail is needed for studies about the causes and impacts of tropical deforestation, for global climate models, for estimating the magnitude of carbon flows and water cycles, and for informing national and international policy and conservation initiatives. For example, programmes such as credits for carbon sequestration will require accurate data on the extent of cover by forest type and the speed of cover change to estimate the value and direction of such credits. Estimates of the rate of carbon sequestration are directly dependent on the information on forest condition because a forest’s capacity to transform atmospheric carbon into biomass is affected not only by forest clearing but also by logging, fragmentation and edge-related tree mortality (Phillips, Malhi, Higuchi, Laurence & Núñez, 1998). Other objectives, such as the identification of global biodiversity conservation priorities, also require additional information about forest cover change patterns, levels of fragmentation and ecosystem integrity (IGBP, 1990, 1992; Kummer & Turner, 1994; Brooks, Pimm & Collar, 1997).

This study focuses on one of the deforestation fronts identified by Myers: the Napo region of western Amazonia, which is shared by Ecuador, Colombia and Peru. The objective is to estimate the extent and rate of deforestation and the level of forest fragmentation for the period 1986–96. The Napo rainforests are among the most biologically diverse and unique environments in the world and have been considered one of the world’s 18 ‘hotspots’ — areas with high biodiversity and under high human pressure (Myers, 1988, 1990; Mittermeier, Myers, Thomsen, Da Fonseca & Olivieri, 1998; Olson & Dinerstein, 1998). Interestingly, previous regional-level studies of forest cover change in the Amazon have focused exclusively on the Brazilian Amazon. Because more than one-third (2.3 million km²) of the Amazon basin (Fearnside, 1990; Skole & Tucker, 1993; Stone, Schlesinger, Houghton & Woodwell, 1994) and two of the three deforestation fronts identified by Myers in this region are not in Brazil, the available information does not provide a complete picture of what is happening to the Amazon rainforests as a whole. Stone et al. (1994) estimated the rainforest area in Bolivia, Brazil, Colombia, Ecuador, Peru and Venezuela at 6.5 million km², including some 0.2 million km² of Pacific lowland forests. A similar estimate is given by Grainger (1993), while Fearnside (1990), Skole and Tucker (1993) and Stone et al. (1994) estimated the Brazilian Amazon rainforest area at 4 million km². An accurate evaluation and understanding of the physical and human dimension of deforestation in the whole of the Amazon basin also needs to take into account rapidly changing areas outside Brazil.

An important contribution of the present work is its evaluation of the extent of forest degradation that accompanies forest fragmentation and how this is linked to land occupation patterns. The spatial patterns of deforestation, and not simply its extent, are critical to the analysis because they affect the capacity of forests to survive and recuperate (Malingreau & Tucker, 1990). Patch size, shape and isolation are key factors defining the potential for recolonization, mortality and dispersion. Hence,
forest fragmentation and border effects make the area affected by deforestation much larger than if it is just measured as forest clearing.

Although national deforestation statistics do exist for all the countries that share the Amazon basin, there are doubts about their reliability and whether they are up to date (with the exception of Brazil). National statistics have intrinsic limitations related to: (a) the lack of information about within-country variability in process and pattern dynamics and an oversimplified biogeographical framework; (b) questionable confidence levels and methodologies and very high variation from source to source; and (c) lack of information about patterns of land cover change and landscape structure. Recent deforestation studies, which are increasingly based on more reliable regional land cover data and the use of satellite imagery, consistently show lower estimates than previously for similar periods, using national surveys or projections. For example, estimates of deforestation in the Brazilian Amazon, the most studied region in the world, during the 1980s and early 1990s show a great variation, with the latest estimates being significantly lower than earlier ones (Fearnside, 1990; Skole & Tucker, 1993; Pedlowski, Dale, Matricardi & Da Silva Filho, 1997; INPE, 1998). In addition, because land cover change processes often span national frontiers, national-level assessments fail to address the problem at an ecosystem level or to identify process characteristics (i.e. rate and extent) for a whole deforestation front. An alternative is to take a regional rather than a country approach, and this is the method used here. This makes it possible to overcome national-boundary restrictions and allows detailed analysis of cover changes by ecosystem type, including forest fragmentation, border impacts and ecosystem integrity assessment.

Research approach

The magnitude, rates and patterns of deforestation were studied for a 100 000 km² area in the Napo region of the western Amazon for the period 1986–96 (Fig. 1). This area is shared by Colombia, Peru and Ecuador (22%, 15% and 63% of the study region, respectively). Regional analysis of land cover characteristics used satellite imagery. Data for 1986 forest coverage were obtained from a mosaic of six Landsat Multispectral Scanner (MSS) images. Data for 1996 were obtained from a mosaic of eight Landsat Thematic Mapper (TM5) images. Areas covered by cloud in one or both periods and not resolved through image mosaics were eliminated from the analysis. Both datasets were registered and converted to a pixel resolution of 100 m. Supervised classification within regions of interest was used to separate among several land cover types. These were later aggregated into three major land cover classes: forest/natural vegetation, deforested and water. Initial land use types within each region of interest were reclassified separately, facilitating a consistent distinction of land use classes and, particularly, the removal of noise in the classification process (i.e. error deforestation pixels appearing in known forested areas). Regions of interest were selected based on extensive fieldwork in the Ecuadorian portion of the study region. (Fieldwork was not carried out on the Colombian side because of increased personal risks due to guerrilla activities.) The level of detail
used in this study, both in terms of resolution and areal extent, allows for a relatively accurate analysis. Larger-scale studies, often using coarse resolution data (e.g. AVHRR), tend to overestimate deforestation by as much as 50% (Skole & Tucker, 1993). More detailed studies may provide more accurate information, but cost per unit of area tends to be much higher, both in personnel and in time requirements. Furthermore, while approximately 10% of the study region was not resolved due to cloud coverage, the deforestation statistics are a close approximation to reality because a significant fraction of the cloud-covered area corresponded to areas known to have full forest coverage.

The definition of deforestation used in this study refers to the complete removal of forest cover for any reason (such as smallholder agriculture or oil extraction). Although some trees may remain, the spectral distinctiveness of these areas is unmistakable. MAG/SUFOREN (n.d.) found that between 10 and 25 standing trees were left per hectare in clearings used for pasture (depending on the size of tree crowns) in the Ecuadorian section of the study region. For this reason, secondary forest in later stages of succession, and other areas of degraded forest, are included in the forest category statistics. Hence, the deforestation estimates presented here should be seen as conservative.

Forest fragmentation analysis allows for diagonal links in patch definition.
Although this level of connectivity underestimates the degree of fragmentation, without it the number of patches exceeded software capabilities (Fragstats cannot process more than 32,000 different patches). Because the region studied includes a large area of natural forests, and because areas of deforestation are concentrated, a more detailed analysis of forest degradation resulting from forest fragmentation was carried out in a subsection of 560,000 ha, where land cover change was found to be dynamic and spanned the period of analysis. In this study, the level of fragmentation was assumed to approximate to the extent of forest degradation in the study region. Forest degradation refers to changes in forest conditions leading to permanent modifications in composition and structure. Forest fragments were grouped by size into various groups denoting a given (qualitative) level of degradation, which is inversely proportional to size. This information is particularly important for estimating the full impact of deforestation on biological resources. This approach is consistent with work done in Peninsular Malaysia, the Philippines and the Brazilian Amazon (respectively: Brown, Iverson & Lugo, 1993; Liu, Iverson & Brown, 1993; Skole & Tucker, 1993).

**Deforestation in the Napo region of the western Amazon, 1986–96**

There are few studies of the dynamics of land cover changes and deforestation in western Amazonia in general, or in the Napo region in particular. Myers (1993, 1994) estimated the area of the western Amazon deforestation front at 200,000 km$^2$ and the area deforested in 1991 alone at 590,000 ha, or an annual deforestation rate of 3%. In the only study found for the Colombian Amazon, Ruiz (1989) estimated that by the late 1980s, 2.5 x 10$^6$ ha of forests had been cleared in the departments of Caquetá, Guaviare and Putumayo (only the department of Putumayo is included in this study). On the Ecuadorian side, Wunder (1997), using official agricultural surveys, estimated that the area deforested in this section of the study region increased from 0.41 x 10$^6$ ha in 1972 to 1.1 x 10$^6$ ha in 1989. This means that 34,705 ha of forests were cleared every year during this period. Interestingly, none of these studies provide maps illustrating the geographic nature of the process.

Land cover change data in this study show that between 1986 and 1996, the area deforested in the Napo region almost doubled, reaching 1.2 x 10$^6$ ha (Table 1). This is equivalent to 12.4% of the study region; up from 6.8% in 1986. On average, 55,590 ha were cleared every year, a rate of 0.6% yr$^{-1}$, which is much lower than Myers’s estimate. All the deforestation occurred in the Colombian and Ecuadorian portions of the study region. Isolation from major urban centres and the lack of roads has hindered forest clearing on the Peruvian side. If the area inside Peru is not taken into account, deforestation rates do not change significantly. Overall, however, deforestation was advancing faster on the Colombian side than on the Ecuadorian side (0.9% yr$^{-1}$ compared with 0.65% yr$^{-1}$). In Ecuador, the deforested area increased from 7.25% in 1986 to 13.4% in 1996, while in Colombia it increased from 10.3% to 18.4%. In terms of area, deforestation estimates for the Ecuadorian portion are roughly half those obtained by Wunder (1997), but the rates are very
Table 1
Forest cover change in the Napo deforestation front of western Amazonia, 1986–96 (in hectares)

<table>
<thead>
<tr>
<th>Land use</th>
<th>1986</th>
<th>1996</th>
<th>Change</th>
<th>Annual change</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>ha</td>
<td>%</td>
<td>ha</td>
<td>%</td>
</tr>
<tr>
<td>Water</td>
<td>116,966</td>
<td>1.2</td>
<td>114,455</td>
<td>1.2</td>
</tr>
<tr>
<td>Forest</td>
<td>9,122,683</td>
<td>92.0</td>
<td>8,569,299</td>
<td>86.4</td>
</tr>
<tr>
<td>Deforested</td>
<td>672,999</td>
<td>6.8</td>
<td>1,228,894</td>
<td>12.4</td>
</tr>
<tr>
<td>Total</td>
<td>9,912,648a</td>
<td>9.122,683</td>
<td>8,569,299</td>
<td>86.4</td>
</tr>
<tr>
<td>Clouds</td>
<td>1,220,932b</td>
<td>6.8</td>
<td>1,220,932b</td>
<td>12.4</td>
</tr>
<tr>
<td>Total study area</td>
<td>11,133,580</td>
<td>9.912,648a</td>
<td>9,569,299</td>
<td>86.4</td>
</tr>
</tbody>
</table>

* Not including areas not resolved due to high cloud coverage.

b Areas covered by clouds in one or both periods.

similar (38,288 ha yr\(^{-1}\) in this study compared with 34,705 ha yr\(^{-1}\) according to Wunder). A comparison could not be made with the Colombian estimates of Ruiz (1989) because of differences in areal coverage and the lack of maps on the original reference.

Figs 2 and 3 show that deforestation is relatively concentrated in the northwest corner of the study region, near the foot of the Andes, from which much of the population is coming. Much deforestation at the local level is tied to the spread of smallholdings and cattle ranching (Eastwood & Pollard, 1992; Vanegas, 1993; Murphy, Bilsborrow & Pichón, 1997; Wunder, 1997). Colonization and development are tied to the expansion of oil extraction and exploration and government programmes, such as colonization schemes and military camps. This is illustrated in Figs 2 and 3 by the presence of clearing axes, the colonization lines, along access roads. In Ecuador, for example, oil exploration and production since the 1970s has opened the region very rapidly through the construction of a large road network. By the 1970s, colonization in other areas of Ecuadorian Amazonia had subsided as the combination of pull forces (oil jobs, increasing access to markets and cheap land) in the north took precedence over those in the south of the Ecuadorian Amazon or on the Pacific side of the Andes (Brown & Sierra, 1994). In Ecuador, the population jumped from 49,578 in 1974 to 82,676 in 1982 and to 103,387 in 1990. Rates of population growth have been around 6–8% yr\(^{-1}\) for several decades, at least twice the national rates. On the Colombian side, population has grown even faster, almost doubling between 1973 and 1985 to reach 119,815. By the last Colombian census, in 1993, the population on the Colombian side of the study area had reached 204,309, an increase of more than 70% in 8 years. On the Peruvian side, the population is still relatively small, confined to a few villages and military posts. (The area studied in Peru falls within the Province of Maynas, which in 1993 had 384,063 people, three-quarters of whom lived in the city of Iquitos, several hundred km outside the study region.)

In addition, land assignations to colonists fostered clearing by imposing titling
conditions that required putting under production at least half the area claimed (Southgate, Sierra & Brown, 1991). Interestingly, deforestation is also evident in areas under indigenous control in both 1986 and 1996. These are along major rivers in the southeastern and central portion of the study region. On the Colombian side, a small fraction of deforestation is also related to the coca economy, as indicated by a large number of small and isolated clearings in the northeastern part of the study region.

Wunder (1997) estimated that 87% of the area cleared in the Ecuadorian portion of the study region was used for cattle ranching by 1989. Vanegas (1993) provides slightly higher estimates for the Colombian portion. This is not surprising, since cattle ranching is a dominant land use in large- and small-scale farms in most of the agricultural frontiers of Latin America. In other parts of the Amazon basin, poor colonists typically allocate 50–65% of the farm area under use to pasture. The remaining area cleared is used to produce subsistence crops, such as rice, cassava, banana and fruit trees, or commercial crops such as coffee and cacao (Loker, 1993; Browder, 1995; Fujisaka, Bell, Hurtado & Crawford, 1996).
Forest fragmentation in the Napo region of the western Amazon, 1986–96

It is often assumed that deforestation spreads as a series of waves, invariably advancing into forested areas. In reality, however, it is not a spatially homogeneous process and its pattern is at least as important as its extent and speed. There is ample evidence that even small levels of fragmentation can cause significant degradation of tropical rainforests. This occurs for several reasons. Tropical populations or key resources (e.g. breeding areas) are often dispersed and the site and time of clearing can determine whether a species or a resource is present or not in a particular forest remnant. Thiollay (1989), for example, found that forest fragments of 10 000 ha of primary forest in French Guiana contained roughly half the number of raptor species found in fragments of 100 000 ha, and that even in these larger fragments not all the expected species were present. Thiollay’s work suggests that compact forest patches of up to 300 000 ha may be needed to ensure the representation of all raptor species in a tropical rainforest region. Clearing also targets specific geomorphological
conditions. Habitats that correlate with these become under-represented in the final mosaic. For example, clearing for agriculture will tend to target well-drained areas (e.g. terrae firm forests) rather than poorly drained areas (e.g. flooded palm forests). Isolation is an additional factor affecting population viability and diversity within each segment as it affects horizontal and vertical population and genetic movements. Work in the Brazilian Amazon shows that interruptions of as little as 80 m can be an important barrier (Bierregaard et al., 1992). Biophysical changes, such as micro-climate and soil modification, also affect forest diversity for at least 100–200 m from the border (Bierregaard & Lovejoy, 1989; Newmark, 1991; Saunders, Hobbs & Margules, 1991; Levey & Stiles, 1992; Kattan, Alvarez-Lopez & Giraldo, 1994).

In general, increased fragmentation affects the carrying capacity of a forest, the sum of fragment areas having a considerably lower capacity than a single patch of the same area. Large animals are among the most vulnerable to forest fragmentation because they need larger areas for survival, have smaller populations and are hunted more often than smaller animals (Terbourgh, 1992). Others are vulnerable because of their dependence on key resources. For example, large insectivore bird extinctions in small fragments have been related to the fast disappearance of army ants after fragmentation. Bierregaard et al. (1992) report vertical biomass changes among small rodents, from greater aerial biomass in continuous forests to greater terrestrial biomass in fragmented forests. This seemed to have been a response to similar changes in insect biomass distribution. Plants are also affected. Terbourgh (1992) has pointed out that the lack of predators and the subsequent increase in seed eaters may favour the expansion of some plant species while reducing others. Bierregaard et al. (1992) show that mortality rates among trees in the edge of fragments of up to 100 ha are almost twice as large as the rates observed in continuous forests in the Amazon basin. At the centre of these fragments, tree recruitment also declines considerably.

Moreover, fragmentation also increases the area exposed to external pressure, primarily from people, which further affects forest structure and composition. Hunting, for example, often eliminates large mammals (including those of great importance for forest regeneration, such as monkeys and rodents) from forest near occupied areas. A brief survey during this study showed no evidence of monkey populations, even in large forest fragments, in the deforestation core of the Ecuadorian portion of the study region. The impact of human pressure on animal resources in this region was studied more formally by Vickers (1988), who found that game populations decreased drastically with increases in hunting pressure. Sierra and Stallings (1998) have shown that selective logging in the lowland rainforests of western Ecuador takes place up to 1 km from forest borders, even in the absence of penetration roads.

Hence, forest fragmentation in the Napo region adds an important dimension to the direct impact resulting from the deforestation discussed earlier. Table 2 shows that by 1986, 1.2% of the remaining forest area was in isolated patches of less than 50 000 ha, almost half being in very small patches of less than 500 ha. By 1996, the area in patches of less than 50 000 ha had tripled, accounting for 3.6% of the remaining forest area. Thus, by 1996, 16% of the forest area in the Napo region can be said to have been either cleared or heavily degraded. This is a conservative estimate. As noted earlier, patches were defined by allowing for diagonal links, which
Table 2
Forest fragmentation in the Napo deforestation front of western Amazonia, 1986–96

<table>
<thead>
<tr>
<th>Patch size (ha)</th>
<th>1986</th>
<th>1996</th>
<th>Change in area (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total area (ha)</td>
<td>% of forest</td>
<td>Total area (ha)</td>
</tr>
<tr>
<td>1–500</td>
<td>58 475</td>
<td>0.6</td>
<td>127 679</td>
</tr>
<tr>
<td>501–1000</td>
<td>4 159</td>
<td>0.0</td>
<td>10 751</td>
</tr>
<tr>
<td>1001–10 000</td>
<td>23 386</td>
<td>0.3</td>
<td>55 367</td>
</tr>
<tr>
<td>10 001–50 000</td>
<td>21 012</td>
<td>0.3</td>
<td>113 628</td>
</tr>
<tr>
<td>1–50 000</td>
<td>107 032</td>
<td>1.2</td>
<td>307 425</td>
</tr>
<tr>
<td>&gt;50 000</td>
<td>9 015 651</td>
<td>98.8</td>
<td>8 261 874</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>9 182 683</td>
<td></td>
<td>8 569 299</td>
</tr>
</tbody>
</table>

greatly exaggerate connectivity among forest fragments. Also, based on the work discussed earlier, it may only be in forest patches upwards of 300 000 ha that forest ecosystems can be preserved in good condition. It is important to recognize that this measure of impact does not discount the extent of forest cover outside the border of the study area, which, if accounted for, may create larger patch sizes. However, because most of the deforestation and fragmentation is rather concentrated, the results of this study do cover most of the area affected, with most of the larger patches generally on the sides and greater fragmentation in the centre.

A closer look and a more strict definition of forest fragmentation (i.e. not allowing for diagonal links) supports this assumption. Fig. 4 illustrates forest cover changes for the Joya de los Sachas area of northwest Ecuador. In 1977, at the beginning of colonization, only 9% of this area had been cleared (not shown). By 1986, the deforested area had increased to 26% and by 1996 to 46%. Forest degradation increased the area affected significantly. In 1977, only 2% of the remaining forest was in fragments of less than 10 000 ha. By 1986, the area of forest in this patch size had increased to 14.5% of the total forest. By 1996, 20 years after colonization had begun, 45% of the forest area was in fragments of less than 10 000 ha. Using 10 000 ha as a threshold patch size, the total forest area affected thus comprised 91% of this study area (Fig. 5). Overall, the rate of deforestation in this subset of the study region was 3.1% yr

The intensification of forest fragmentation in the Napo region of western Amazonia is directly related to the land occupation patterns discussed earlier. A detailed study of farm households in the Joya de Los Sachas area, by Pichón and Bilsborrow (1992, cited in Murphy et al., 1997) found that the region was dominated by smallholdings. Over 90% of the farms were smaller than 60 ha, with an average of 44 ha and a range from 1.15 to 400 ha. The only large-scale agricultural developments in this region, not included in the Pichón and Bilsborrow study, are two oil palm plantations, each of approximately 10 000 ha. One of these is visible in Fig. 4 as a large compact cleared area in the southeast corner.

The expansion of agricultural activities by these farms forms a specific pattern
Fig. 4. Deforestation and forest fragmentation in La Joya de Los Sachas area, 1986 and 1996.
Fig. 5. Remaining forest in selected patch sizes in La Joya de Los Sachas area, 1986 and 1996 (%).
Conclusions

This study shows that deforestation in the Napo region of western Amazonia has affected a significant area but that its extent and the speed at which it is increasing is less than previously reported. Myers’s (1993) estimate of 590,000 ha of deforestation per year in western Amazonia, although for an area twice as large as that examined here, seems to be a significant overestimation. Deforestation in the region probably stands at 100,000–200,000 ha yr\(^{-1}\), an annual rate of less than 1%. This is considerably lower than the 3% rate reported by Myers, and is consistent with findings in other areas of the Amazon, where recent estimates of deforestation rates based on satellite information are below 1%, which is also lower than earlier estimates.

Results also suggest that available national-level statistics overestimate deforestation. In Ecuador, for example, recent estimates put deforestation in the period 1990–5 at roughly 190,000 ha yr\(^{-1}\), an annual rate of 1.6% (WRI, 1998). The present study found that in the north of the Ecuadorian Amazon, one of the two main deforestation fronts in the country, approximately 40,000 ha were cleared a year in the period 1986–96, at an annual rate of 0.65%. An earlier study for northwest Ecuador, the other major deforestation front in Ecuador, found that in the period 1983–93, approximately 10,000 ha of natural forests were cleared every year. In this region, however, due to the small remnant forest area, the annual deforestation rate stood at 1.9% for this period (Sierra & Stallings, 1998). If it is assumed that one-third of the deforestation in Ecuador occurs in these two deforestation fronts, the area of natural forest cleared in Ecuador every year would be approximately 150,000 ha, or three-quarters of the previous estimate. The rate would also be at levels below 1–1.2% yr\(^{-1}\), as opposed to 1.6% yr\(^{-1}\). Because the area studied here that falls within Colombia and Peru is not representative of the process at a national scale, a similar assumption cannot be made for these countries. Locally, however, rates of deforestation may be much faster. In the Joya de los Sachas area, deforestation was very close to Myers’s estimate. Clearly, estimates of rates and extent are affected by the size of the area studied and the level of detail. The smaller the area studied, the more extreme these measures are likely to be.

No comfort, however, can be derived from these comparisons. Even at these lower rates deforestation is significant. Furthermore, its overall impact is in fact much greater than that suggested by conventional statistics such as extent and rates. Using the results from the Joya de los Sachas region as a guide, it is reasonable to argue that by the time 50% of the forest cover has been cleared, most of the remaining forest has also lost critical resources for conservation in general, and for biodiversity conservation in particular. By then, a significant fraction of the remaining forest consists of many small forest remnants. This is especially important for conservation planners who need to consider that as time passes forest fragmentation intensifies more than proportionally in relation to cleared area. Beyond a hitherto undefined threshold, even very large aggregate measures of forest cover do not provide a good basis for biodiversity conservation. At this point it can be expected that the aggregate forest area will comprise a large number of small forest patches. Designing and creating new reserves of sufficient size, for example, would be very difficult once that threshold has passed.
Finally, a critical implication of this work is that deforestation and forest degradation in the Amazon region outside Brazil is significant and merits study in its own right. If similar deforestation patterns are assumed for the rest of non-Brazilian Amazon, the area deforested in the whole region by 1996 would be approximately 285,000 km² (12.39% of 2.3 million km²). While this study does not provide direct support for such an assumption, there is evidence that the occupation process in the Napo region is representative of processes in large areas of the western Amazon. Loker (1993), for example, reports similar farm sizes and Aramburu (1984) describes a similar occupation process for the Peruvian Amazon. Vanegas (1993) does the same for the Colombian Amazon.

Hence, deforestation and forest fragmentation in the non-Brazilian Amazon may be as critical as in the Brazilian portion. The latest and probably most accurate and comprehensive study of deforestation in the Brazilian Amazon (INPE, 1998) found that by 1996, 517,000 km² had been converted to agriculture or some type of non-forest land use. This area, however, includes clearing of non-rainforest formations (e.g. cerrado) and it is unclear how much of it applies specifically to the Amazon rainforest. A reasonably low estimate is provided by Skole and Tucker (1993), who found that by 1988 the total area deforested in the Brazilian Amazon was 230,000 km², equivalent to 5.7% of the original forest area. Higher estimates are provided by Fearnside (1990), who estimated the area deforested by 1988 at 348,000 km², and Mahar (1989), who estimated 598,922 km². If the estimates of Skole and Tucker and Fearnside are used as a starting point to calculate lower and upper estimates, respectively, of the deforestation in the Brazilian Amazon proper in 1988, and the annual deforestation rates estimated by INPE (1998) for the period 1988–96 are applied on a year-by-year basis, a low estimation of the deforested area in the Brazilian Amazon by 1996 would be 370,000 km², or 9.25% of the original forest extent, and a high estimate would be 482,200 km², or 12% of the original forest extent. Hence, proportionally, the level of deforestation in these two Amazon regions may be very close and the environmental risks similar. Overall, it seems reasonable to assume that at least one-third of the deforestation in the Amazon occurs outside Brazil.

It is not possible to provide a better appraisal of this and other issues addressed here for complete ecoregions, such as the whole Amazon rainforests or the rainforests on the western sides of the Andes, or at a global scale, until a representative number of areas are studied in sufficient detail. Such regional coverage is also important because it would provide the land cover and land cover change data that has been considered a top priority for global environmental research (IGBP 1990, 1992; Malingreau, Achard, D’Souza, Stibig & D’Souza, 1995). The definition of deforestation as a local, regional or even global problem depends on its magnitude and spatial distribution (Malingreau & Tucker, 1990). For these reasons, more geographically diverse regional studies are also necessary to grasp the complexity of environmental processes and to understand more about people–environment interactions.

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