2 Spatial and Temporal Patterns of Forest Loss and Fragmentation in Mexico and Chile

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Fragmented forest landscape in the Highlands of Chiapas, Mexico. Photo: Luis Cayuela

Summary

The patterns and driving forces of forest loss and fragmentation were assessed in four study areas: two in Mexico (Central Veracruz and the Highlands of Chiapas) and two in Chile (Rio Maule-Cobquecura and Los Muermos-Ancud). For the Highlands of Chiapas, Rio Maule-Cobquecura and Los Muermos-Ancud study areas, three land-cover maps were derived from satellite imagery acquired between 1975–1976 and 1999–2000. For Central Veracruz, two land-cover maps were obtained from the interpretation of aerial photographs and Landsat ETM+ satellite images for 1984 and 2000, respectively. Analysis of these images indicated a reduction in natural forest area of 67% in Rio Maule-Cobquecura, 57% in the Highlands of Chiapas, 26% in Central Veracruz and 23% in Los Muermos-Ancud. These losses are equivalent to annual forest loss rates of 4.4%, 3.4%, 2.0% and 1.1% per year, respectively. Forest fragmentation in the study areas led to a decrease in forest patch size, which was associated with a rapid increase in the density and isolation of forest patches and a decline in area of interior forests and number of large patches. Logistic regression models were used in each study area to identify the factors associated with forest loss. Overall, the probability of an area being cleared of forest was greatest in gently sloping areas and around the margins of forest patches. Additionally, soil fertility appears to be a significant factor associated with deforestation in Central Veracruz. In Maule-Cobquecura and Los Muermos-Ancud the probability of deforestation was higher as size of forest fragments decreased, whereas in the Highlands of Chiapas large fragments were particularly vulnerable to deforestation. Given the current trends of forest loss, we predict that further declines and spatial changes of forest cover will occur in each of the study areas. The patterns observed reveal some of the immediate causes of deforestation in Mexico and Chile such as pasture and crop expansion, forest logging and conversion to plantations of exotic tree species. These changes highlight some weaknesses in the national environmental and economic policies in the countries included in this study.

Introduction

Habitat fragmentation and forest loss have been recognized as major threats to ecosystems worldwide (Iida and Nakashizuka, 1995; Dale and Pearson, 1997; Noss, 2001; Armenteras *et al.*, 2003). These two processes have negative effects on biodiversity by increasing isolation of habitats (Debinski and Holt, 2000), reducing the extent of species habitat and modifying the population dynamics of species (Watson *et al.*, 2004). Fragmentation may also have negative effects on species richness by reducing the probability of successful dispersal and establishment (Gigord *et al.*, 1999) as well as by reducing the capacity of a patch of habitat to sustain a resident population (Iida and Nakashizuka, 1995). For example, fragmentation of Maulino temperate forest in central Chile has affected the abundance of bird richness (Vergara and Simonetti, 2004) and regeneration of shade-tolerant species (Bustamante and Castor, 1998), and has also favoured the invasion of alien species (Bustamante *et al.*, 2003). The ecological consequences of fragmentation can differ, depending on the pattern or spatial configuration imposed on a landscape and how this varies both temporally and spatially (Ite and Adams, 1998; Armenteras *et al.*, 2003). Some studies have shown that the spatial configuration of the landscape and community structure may significantly affect species richness at different scales (Steiner and Köhler, 2003). Other authors emphasize the need to incorporate the spatial configuration and connectivity attributes at a landscape level in order to protect the ecological integrity of species assemblages (Herrmann *et al.*, 2005; Piessens *et al.*, 2005).

The temporal evaluation of forest change based on satellite imagery linked to fragmentation analysis is becoming a valuable set of techniques for assessing the degree of threat to forest ecosystems (Luque, 2000; Franklin, 2001; Imbernon and Branthomme, 2001; Sader *et al.*, 2001; Armenteras *et al.*, 2003). A number of imagery-based studies of deforestation have been conducted in tropical forests (Skole and Tucker, 1993; Turner and Corlett, 1996; Imbernon and Branthomme, 2001; Sader *et al.*, 2001; Steininger *et al.*, 2001; Laurance *et al.*, 2006), including the Amazon (Jorge and García, 1997; Pedlowski *et al.*, 1997; Ranta *et al.*, 1998; Laurance, 1999; Laurance *et al.*, 2000; Sierra, 2000), but few studies of deforestation and fragmentation have been made in temperate forests (Gibson *et al.*, 1988; Staus *et al.*, 2002; Hobbs and Yates, 2003), or in tropical montane forests.

There is a global need to identify the causes of deforestation and forest fragmentation and to understand how these affect the spatial configuration of landscapes over time (Angelsen and Kaimowitz, 1999; Verburg *et al.*, 2002; Bürgi *et al.*, 2004; McConnell *et al.*, 2004; Veldkamp and Verburg, 2004). It is increasingly recognized that simple descriptions of land-cover types are inadequate for conservation planning or resource management, because they do not incorporate information about the patterns of land-use change that can have profound effects on ecological process of interest (Bürgi *et al.*, 2004; Corney *et al.*, 2004). For a more systematic understanding of landscape change, it is necessary to study the driving forces responsible for deforestation, leading to the analysis of processes and not merely patterns (Bürgi *et al.*, 2004). The ability to link a particular driver in the landscape to specific landscape changes is a powerful tool for researchers exploring environmental change (Evans and Moran, 2002). Some researchers add that it is necessary to move beyond the simplistic assessment of the proximate causes of land-use and land-cover change and assess underlying factors such as environment–development policies (Lambin *et al.*, 2001; Silva, 2004).

In this chapter we examine the rates and patterns of deforestation and fragmentation of native forests in each of four study areas in Mexico and Chile. In addition, we analyse the influence of social and environmental driving forces on landscape change in each study area. During the past three decades, expansion of croplands, pasturelands and industrial plantations has resulted in a substantial decline in forest area and in an increase in forest fragmentation. Some research on the ecological consequences of forest fragmentation has previously been undertaken in Chile and Mexico (Willson *et al.*, 1994; Bustamante and Grez, 1995; Donoso *et al.*, 2003; Vergara and Simonetti, 2004; Martínez-Morales, 2005), but few studies have integrated spatial and temporal analyses to assess the pattern and rate of forest loss and fragmentation.

Methods

Study areas

Four study areas were selected from southern Mexico and south-central Chile (Fig. 2.1): (a) the central part of Veracruz in Mexico (Central Veracruz); (b) the

Fig. 2.1. Location of the four study areas in Mexico and Chile.

highlands of the state of Chiapas in Mexico (the Highlands of Chiapas); (c) the Coastal Range in Chile, from Rio Maule (VII region) to Cobquecura municipality in the VIII region (Rio Maule-Cobquecura); and (d) the area situated from Los Muermos to the Chiloé Island in Chile, including the entire municipality of Ancud (Los Muermos-Ancud). These study areas were selected to contrast the effects of different historical patterns of deforestation and different human pressures on the forest ecosystems.

Cloud forests are important to study for a number of reasons. Cloud forest covers less than 1% of the total area of Mexico, yet contains around 2500–3000 plant species, representing about 10–12% of the total number of plant species that occur in Mexico (Rzedowski, 1993; Mitermeier *et al.*, 1997). Moreover, cloud forests have the highest number of mammal species (95) of any type of forest in Mexico and a high rate of endemism in plants (30%) (Ramamoorthy *et al.*, 1993). Despite this high biodiversity, more than 50% of the area of this type of forest has already been converted to other land uses nationally (Challenger, 1998).

The Highlands of Chiapas are also a biologically diverse region, extending over 11,000 km^2, and include 30% of about 9000 vascular plant species of the flora of Chiapas (Breedlove, 1981). Several forest formations are found in the Highlands, including oak, pine–oak, pine and montane cloud forests (Miranda, 1952; Rzedowski, 1978; González-Espinosa *et al.*, 1991). Our study area covers *c.*3550 km^2, with elevation ranging from 600 to 2900 m (mostly above 1500 m). The topography is abrupt, with fairly steep slopes. The climate is cool and humid, with a rainy summer. The region is densely populated by Mayan peasants, who have cleared the forest for shifting cultivation and extracted firewood and other forest resources since pre-Columbian times (Collier, 1975). The main economic activities are traditional agriculture and non-commercial forestry. Slash-and-burn agriculture and long-term exploitation of forests for fuelwood have contributed to the expansion of relatively low diversity pine and mixed pine–oak stands coupled with a reduction in the extent of highly diverse oak and mountain cloud forests (Ramírez-Marcial *et al.*, 2001; Galindo-Jaimes *et al.*, 2002). The cloud forests in Central Veracruz are also highly threatened as a result of deforestation and urban expansion (Williams-Linera *et al.*, 2002). The study area covers *c.*7166 km^2 of the mountainous region (> 800 m) in the centre of the state.

The two Chilean study areas are located in the temperate forest zone, which has been classified as a biodiversity hotspot for conservation (Myers *et al.*, 2000) and has also been included among the most threatened eco-regions in the world in the Global 200 initiative launched by WWF and the World Bank (Dinerstein *et al.*, 1995). The Rio Maule-Cobquecura study area covers *c.*5781 km^2. The natural forest is mainly dominated by secondary forest of *Nothofagus* species (*N. obliqua* and *N. glauca*) (Fagaceae) and sclerophyllous species. At present, approximately 5% of the native forest in the VII region is under the National System of Protected Areas (SNASPE), while the remaining forests lack cohesive protection.

Los Muermos-Ancud covers *c.*5032 km^2 and is characterized by a rainy temperate climate with an oceanic influence and without dry periods (Di Castri and Hajek, 1976). The landscape is dominated by a broadleaved evergreen temperate rainforest within a matrix of agricultural land and pastures. In the middle of the 20th century a significant area of native forests was cut down and burnt as a result of European settlement. Intensive timber exploitation then began in the area, allowing the establishment of areas for grazing and crop cultivation (Donoso and Lara, 1995). In Chiloé Island, the process of deforestation by logging and cultivation occurred mainly in recent decades, its exploitation having been delayed by virtue of its isolation from the mainland.

Generation of spatial data

To analyse the spatial and temporal patterns of land-cover change we used Landsat satellite scenes and aerial photographs, along with the geographic information system (GIS). In Central Veracruz, the analysis of land-cover changes was conducted over a 16-year period using thematic maps generated for the years 1984 and 2000 (Palacio-Prieto *et al.*, 2000). The 1984 coverage was based on digitized topographic maps generated from black and white aerial photographs (INEGI, 1984) and the 2000 map was obtained from the National Forest Inventory (Palacio-Prieto *et al.*, 2000) using Landsat ETM+ (Enhanced Thematic Mapper) satellite images (November 1999 to May 2000).

For the Highlands of Chiapas, Rio Maule-Cobquecura and Los Muermos-Ancud, a set of three satellite images were acquired at different time intervals over the last three decades (Table 2.1). Similar to the methodology described by Fuller (2001), Hansen *et al.* (2001) and Staus *et al.* (2002), the original 79 m MSS (Multispectral Scanner) raster grids were resampled to the resolution of the TM (Thematic Mapper) and ETM+ raster grids (30 m). The resampling enabled the land-cover maps to be produced with consistent resolution, which is essential to develop meaningful comparisons between scenes from different dates. Each image was geometrically, atmospherically and topographically corrected. For the Highlands of Chiapas, the classification of satellite imagery was undertaken applying the Dempster–Shafer theory of evidence (Shafer, 1976), which enabled an increase in the accuracy of classification by the combination of remote sensing data with information derived from expert knowledge (Cayuela *et al.*, 2006a). Classifications of the land-cover types in Chile were conducted using the decision criterion of Maximum Likelihood (Chuvieco, 1996) and set of points of field visits and thematic maps developed by a comprehensive cartographic study of natural vegetation known as Catastro (CONAF *et al.*, 1999).

In Central Veracruz, the 1984 (72 vegetation classes) and 2000 (43 classes) land covers were reclassified using six categories (crop and pasture land, forest, old-field, native forest, urban areas and other) to facilitate comparisons and simplify the evaluation of patterns of land-use change. For this reclassification, disturbed forest (including herbaceous and shrub cover) was grouped into 'old-field', while all other natural vegetation cover types were

Table 2.1. Spatial data used in each study area.

Study area	Type of data	Year
Central Veracruz	Aerial photographs	1984
	Landsat 7 – ETM+	1999–2000
The Highlands of Chiapas	Landsat 1 – MSS	1975
	Landsat 5 – TM	1990
	Landsat 7 – ETM+	2000
Rio Maule-Cobquecura	Landsat 1 – MSS	1975
	Landsat 5 – TM	1990
	Landsat 7 – ETM+	2000
Los Muermos-Ancud	Landsat 1 – MSS	1976
	Landsat 5 – TM	1985
	Landsat 7 – ETM+	1999

MSS, Multispectral Scanner; TM, Thematic Mapper; ETM+, Enhanced Thematic Mapper.

grouped into the 'other' category. In the Highlands of Chiapas, three land-cover classes were defined: (i) native forest, including pine, pine–oak, oak and montane cloud forests; (ii) shade coffee plantations; and (iii) non-forest cover, which corresponded to agriculture fields, pasture lands, recent fallows, cleared areas, bare ground and urban areas (Cayuela *et al.*, 2006b). In the Chilean study areas, the following main categories of land-cover types were obtained: crops and pasture lands, shrublands, arboreous shrublands, native forests, and other land-cover types. In Rio Maule-Cobquecura, native forests corresponded basically to secondary forests, whereas in Los Muermos-Ancud this category included secondary and old-growth forests.

Analyses of forest loss and landscape spatial pattern

The resulting categories of land-cover types were used to analyse forest cover change over time using GIS software. Forest maps for each study year were generated to quantify forest loss and the spatial configuration of native forest fragments. The formula used to determine the annual rate of deforestation was (FAO, 1995):

$$P=\left[\left(\frac{A_2}{A_1}\right)^{1/(t_2-t_1)}-1\right]*100$$

where P is the percentage loss per year, A_1 and A_2 are the forest area at time t_1 and t_2 respectively.

Next, landscape spatial indices were computed using FRAGSTATS (version 3.3) (McGarigal *et al.*, 2002). The following indices were calculated: (i) mean patch size (ha); (ii) patch density (number of patches per 100 ha); (iii) the largest patch index (percentage of area accounted for by the largest patch); (iv) the total edge length (km); (v) total core area (interior area remaining after removing an edge depth of 100 m, in hectares); and (vi) mean proximity index (ratio between the size and proximity of all patches whose edges are within a 1-km search radius of the focal patch). In Central Veracruz, the forest loss analysis was conducted using only undisturbed forest and excluding disturbed forest or old-fields. In the other study areas, forest loss was determined using a unique category of native forest.

Determination of driving forces of deforestation

The question of which environmental and social factors ('drivers') affected the probability of a particular location being deforested was investigated by logical regression analyses. Cover maps from consecutive images (e.g. 1976 and 1985 in Los Muermos-Ancud) were overlain in a GIS, and each pixel of the image was classified as either forested (i.e. forest in both years) or deforested (i.e. forested in the first year and some other cover type in the second year). A random subset of 1000 forested-plus-deforested pixels was then selected from each study area, the pixels being chosen so that the distance between them was at least 1500 m, in order to reduce the degree of spatial autocorrelation within the data. Models were fitted using logistic regression (Crawley, 2005), with a binary response variable ('0' for forested pixels, '1' for deforested pixels), a logit link function and a linear combination of explanatory variables. All the explanatory and response variables used in the analysis were based on pixel sizes of 30 m × 30 m.

Many explanatory variables were available for each site (see below). In the first round of modelling, we tested whether explanatory variables affected deforestation probability by fitting a series of univariate models, and testing the statistical significance of each variable using a χ^2 test. Next, all variables deemed to be significant by this approach ($P < 0.05$) were entered into a multivariate analysis, the purpose of which was to test whether the list of significant factors could be reduced because of covariance among variables. A backward selection method was used to test whether the change in deviance associated with dropping terms out of the model was statistically significant (χ^2 test), until a 'minimum adequate model' was produced in which all terms were significant at $P < 0.001$.

In Central Veracruz, a total of 15 continuous variables, derived from raster maps (100 m^2 pixels), were included as explanatory variables in the logistic regression analysis. These factors included elevation (m), slope (°), soil fertility (range 1–4 with 0.5 increments), mean annual precipitation (mm), latitude (0.5° increments), distance to roads, rivers and agricultural fields (m), as well as road, river and agriculture density (% pixels) in a 2 km radius, distance to national parks (m), distance to initial forest edge (m), population density (people km^{-2}) and population growth rates (% change). Precipitation, road, river, elevation and slope data were provided by Mexico's National Water Commission (CNA; scale 1:250,000), with the latter two variables based on a digital elevation model generated from topographic maps (1:50,000) with 50 m elevation increments. Soil coverage obtained from the National Commission for the Knowledge and Use of Biodiversity (CONABIO; modified from INIFAP, 1995) was assigned to soil fertility values by a geomorphologist who is familiar with the region (D. Geissert, Instituto de Ecología, Xalapa, Veracruz, Mexico, 2005, personal communication). A map of national parks was also obtained from the CONABIO geographic database (CONANP, 2003). Finally, population data were obtained from the 1995 and 2000 National Censuses (INEGI, 1995, 2000). In the Highlands of Chiapas, elevation (m) and slope (°) were extracted from a 1:50,000 digital elevation model. An index of soil fertility/quality following González-Espinosa *et al.* (2004) was generated based on the interpretation of physical and chemical properties of soil taxa described by FAO–UNESCO maps (Duchaufour, 1987). Road density (m km^{-2}) was calculated using a 1:50,000 digitized road map, giving different weights to paved and unpaved roads. Human population density was obtained by dividing the study area according to the location of human settlements into a meaningful tessellation of Thiessen polygons. Population density was then calculated by dividing total population in each settlement (INEGI, 2000) by the area of its corresponding polygon.

In the Chilean study areas, grid maps for slope (°), elevation (m), distance to roads (km), distance to rivers (km) and distance to urban areas (km) were generated using data set at a scale of 1:50,000 of the National Vegetation Mapping (CONAF *et al.*, 1999). Soil types were acquired at a scale of 1:250,000 (Schlatter *et al.*, 1995).

Road, urban and human population variables were used as surrogate measures of human pressure for all study areas, except for Rio Maule-Cobquecura and Los Muermos-Ancud, where population density is relatively low. Maps of patch size, non-forest density and distance to patch edge were calculated using the forest cover maps derived for each study area. Soil variables were available as categorical variables and the remainder as continuous variables. These data sets are the most comprehensive presently available for the study areas. It is assumed that the contribution of socio-economic and environmental variables to deforestation have operated from 1975–1976 to 1999–2000, and will continue to operate over the next few decades.

Results

Forest loss and land-cover change

Central Veracruz

From 1984 to 2000, the percentage of the landscape represented by crop and pasture land declined from 59.6% to 50.2% (Table 2.2, Fig. 2.2). The total forest area increased from 273,251 ha to 329,908 ha, representing 38.1% and 46% of the landscape, respectively. However, the loss of undisturbed native forest was equivalent to a deforestation rate of 2.0% per year. Conversely, the area of old-fields and disturbed secondary forest increased from 8.7% to 24.2%. Urban areas, included in other categories of land-cover types, showed a substantial increase from 2.3% of the landscape to 3.8%.

The Highlands of Chiapas

Crop and pasture lands covered 18% of the total study area in 1975 (Table 2.2, Fig. 2.3), increasing substantially to 39% in 1990 and to 57% in 2000. An opposite trend was observed for coffee plantations: they declined from 8% in 1975 to 4% in 2000. In 1975, the native forests covered approximately two-thirds of the area of the study landscape. Twenty-five years later, this figure declined to less than one-third. This is equivalent to a total forest loss of 57% between 1975 and 2000, and to a deforestation rate of 3.4% per year for the entire study period. However, there were differences between time intervals. Between 1975 and 1990 the forest loss rate was 1.5% per year, whereas between 1990 and 2000 this rate increased considerably to 6.1% per year.

Rio Maule-Cobquecura

Crop and pasture lands slightly increased during the entire study period, from 18% to 22% (Table 2.2, Fig. 2.4). The shrublands and arboreous shrublands comprised 54% of the landscape in 1975; 25 years later, these land-cover types

Table 2.2. Estimates of area of land-cover types, in hectares and percentage of total classified area, in the four study areas. The land-cover type named as 'other categories' includes urban areas, bare ground and other types of natural vegetation.

Central Veracruz, Mexico

	1984		2000	
Land-cover type	ha	%	ha	%
Crop and pasture land	426,877	59.6	359,631	50.2
Native forest*	273,251	38.1	329,908	46.0
Urban areas and other categories	16,493	2.3	27,081	3.8
Total	716,621	100.0	716,620	100.0

*Includes old-fields (disturbed secondary forests).

The Highlands of Chiapas, Mexico

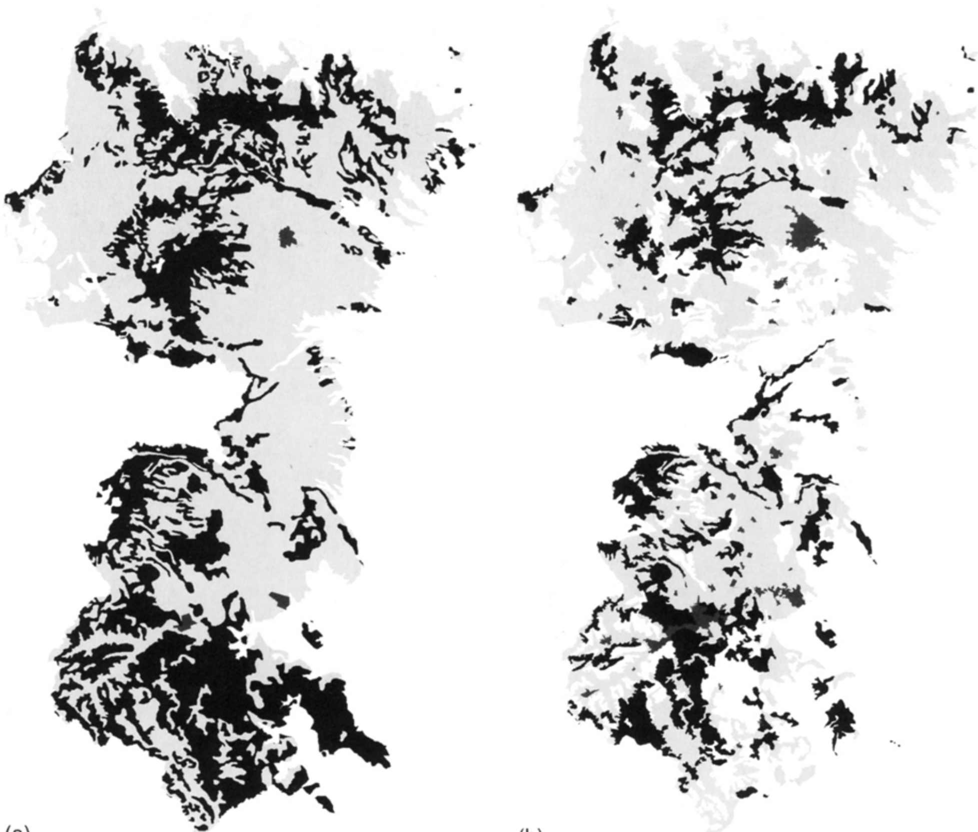
	1975		1990		2000	
Land-cover type	ha	%	ha	%	ha	%
Crop and pasture land	61,346	17.5	134,579	38.8	193,915	56.8
Coffee plantation	27,689	7.9	19,036	5.5	15,010	4.4
Native forest	231,605	66.0	183,501	52.9	98,339	28.8
Other categories	30,044	8.6	9,811	2.8	34,006	10.0
Total	350,684	100.0	346,927	100.0	341,270	100.0

Rio Maule-Cobquecura, Chile

	1975		1990		2000	
Land-cover type	ha	%	ha	%	ha	%
Crop and pasture land	105,701	18.3	78,482	13.6	124,819	21.6
Shrubland	193,532	33.5	260,607	45.1	104,151	18.0
Arboreous shrubland	112,818	19.5	79,643	13.8	93,261	16.1
Native forest	119,994	20.8	56,133	9.7	39,002	6.7
Exotic species plantation	29,579	5.1	96,777	16.7	211,686	36.6
Other categories	16,541	2.9	6,522	1.1	4,800	0.8
Total	578,164	100.0	578,164	100.0	578,164	100.0

Los Muermos-Ancud, Chile

	1976		1985		1999	
Land-cover type	ha	%	ha	%	ha	%
Crop and pasture lands	46,643	9.3	120,008	23.8	129,008	25.6
Shrubland	101,902	20.2	53,270	10.6	34,642	6.9
Arboreous shrubland	78,349	15.6	66,697	13.3	95,113	18.9
Native forest	266,852	53.0	230,410	45.8	206,736	41.1
Other categories	9,541	1.9	32,902	6.5	37,788	7.5
Total	503,287	100.0	503,287	100.0	503,287	100.0



(a) (b)

Fig. 2.2. Major land-cover types in Central Veracruz for the years (a) 1984 and (b) 2000. Light grey, crop and pasture land; black, native forest; medium grey, urban areas.

comprised 34% of the total area. The exotic-species plantations increased from 5% in 1975 to 17% in 1990; by 2000 this land-cover type was the dominant vegetation type on the map, comprising 37% of the land area. During the whole study period, the estimated cover of native forests decreased from 119,994 ha in 1975 to 39,002 ha in 2000. In other words, 67% of the native forest existing in 1975 had disappeared by 2000, which was equivalent to an annual deforestation rate of 4.4% per year. Most of the forest loss was concentrated in the first 15 years of the study period, at a deforestation rate of 5.0% per year. Between 1990 and 2000, the rate decreased slightly to approximately 3.6% per year. Throughout the study period, more than half (53%) of the native forests existing in 1975 had gradually been converted into exotic-species plantations by 2000; another substantial area (40% of native forest in 1975) was transformed into shrublands or arboreous shrublands.

Los Muermos-Ancud

Crop and pasture lands substantially increased during the first time interval, from 9% to 24% (Table 2.2, Fig. 2.5). During the second period of analy-

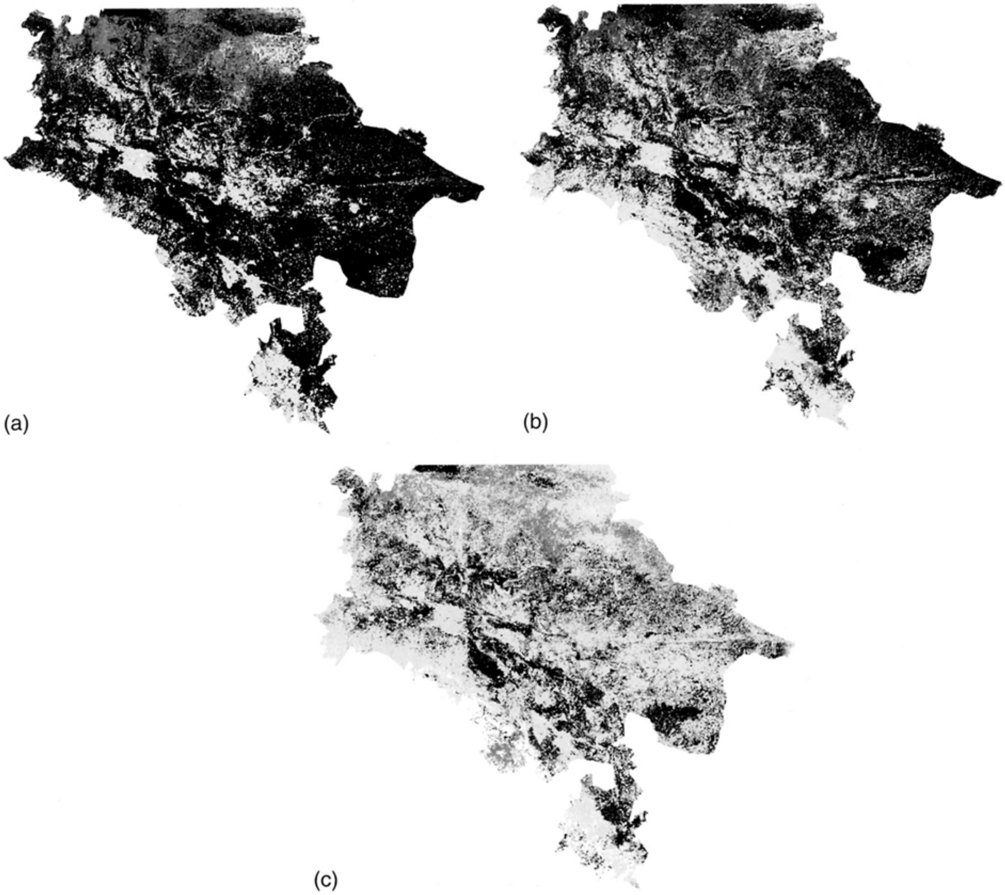


Fig. 2.3. Major land-cover types in the Highlands of Chiapas for the years (a) 1975, (b) 1990 and (c) 2000. Light grey, crop and pasture; black, native forest; medium grey, coffee plantation.

sis, this cover was relatively stable, representing about 26% of the study area. Between 1976 and 1985, the total area of shrublands and arboreous shrublands decreased from 36% to 24%. This decrease was followed by a subsequent increase to 27% in 1999. The estimated area of native forests decreased from 266,852 ha in 1976 to 206,736 ha in 1999, equivalent to 53% and 41% of the total classified area respectively. This means that approximately 23% of the native forests in 1976 had disappeared by 1999, at an annual forest loss of 1.1% per year. Most of the forest loss was concentrated in the first 9 years of the study period, at a deforestation rate of 1.6% per year. In the second time interval, this rate decreased to 0.8% per year. During the time intervals, 29% of the native forests were replaced by arboreous shrublands and 8% by shrublands. The loss of native forests has been associated with an increasing proportion of arboreous shrublands, and also to an increase in the area of crop and pasture lands.

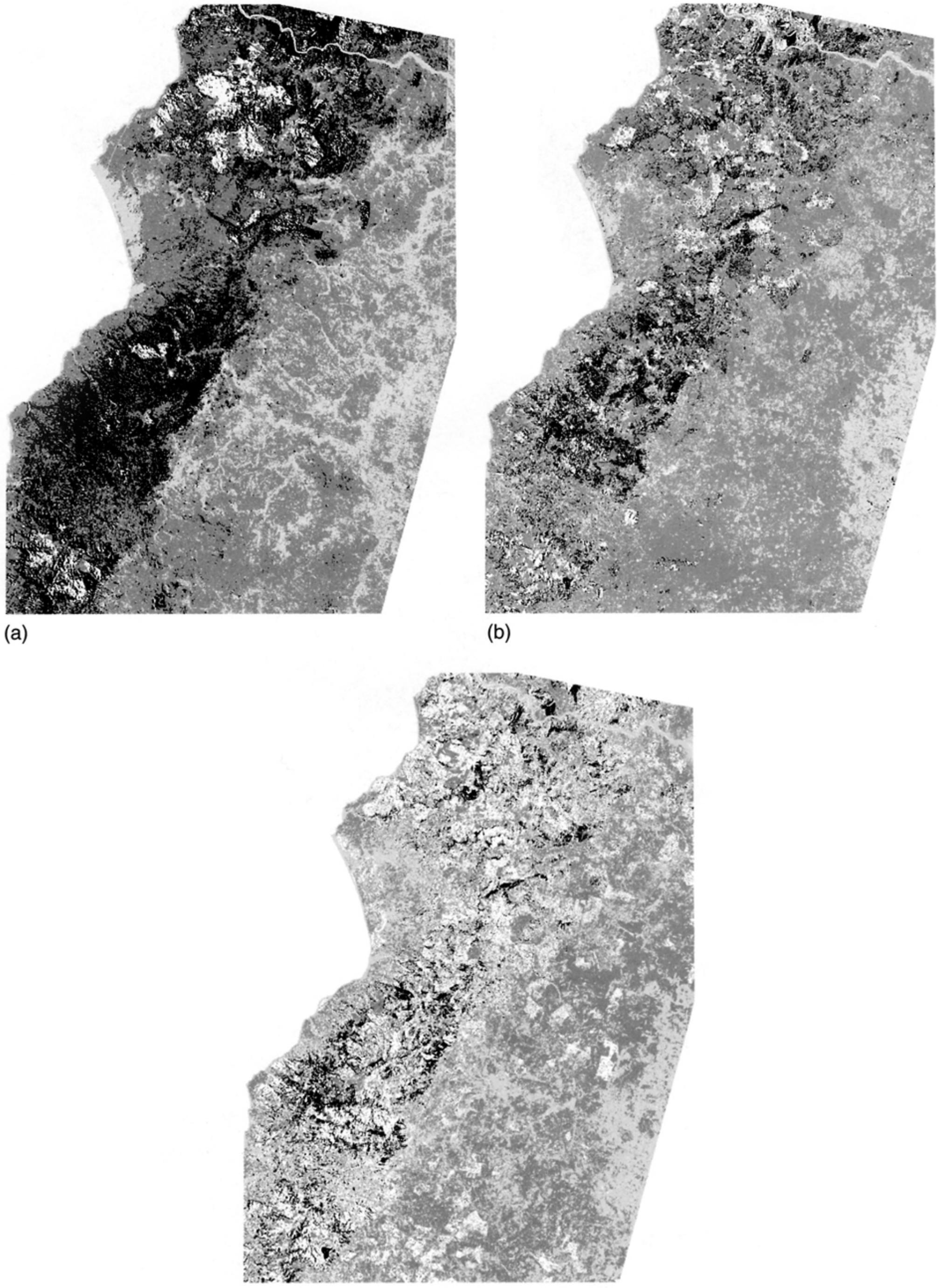


Fig. 2.4. Major land-cover types in Rio Maule-Cobquecura for the years (a) 1975, (b) 1990 and (c) 2000. Light grey, crop and pasture land; medium grey, shrubland and arboreous shrubland; black, native forest; white, exotic species plantation.

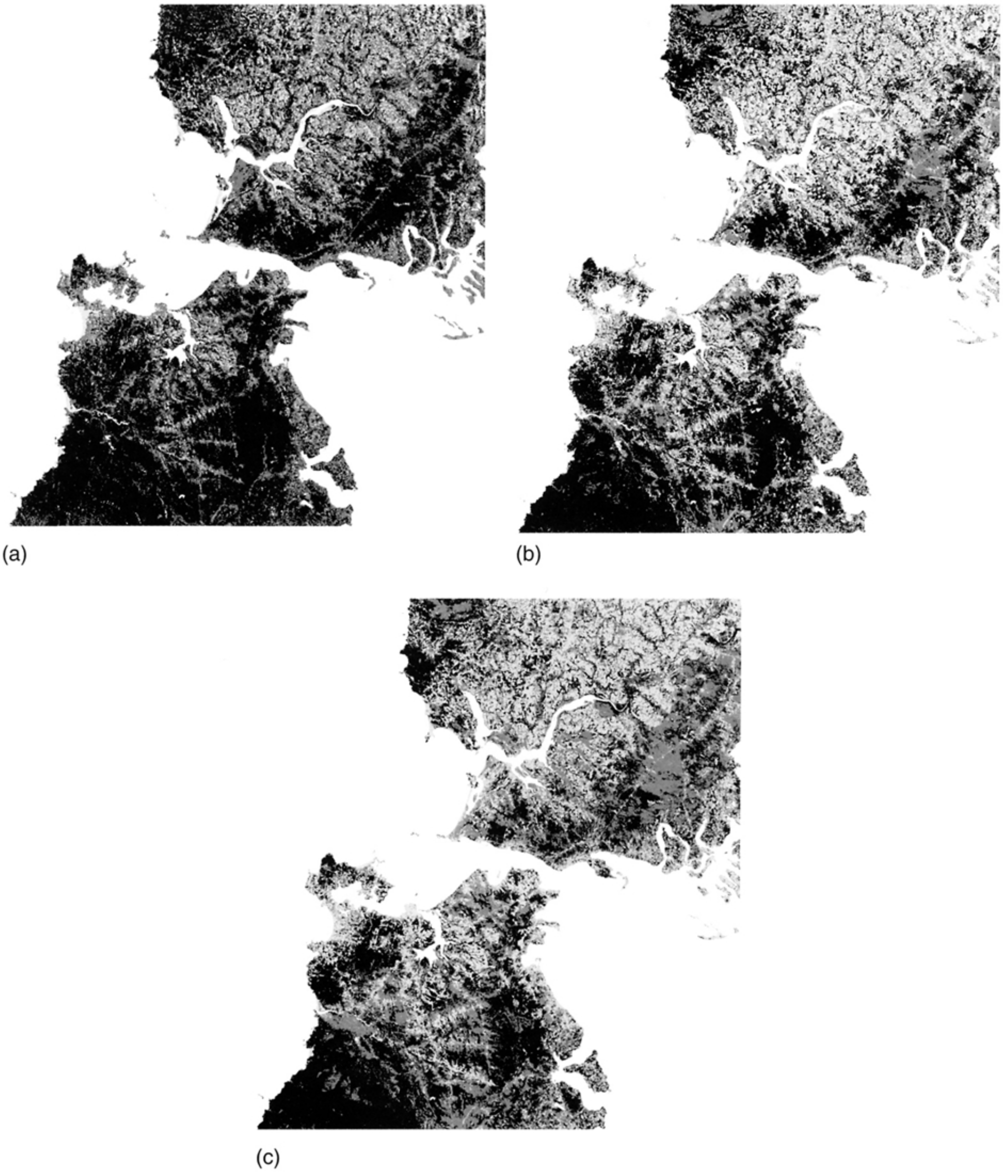


Fig. 2.5. Major land-cover types in Los Muermos-Ancud for the years (a) 1976, (b) 1985 and (c) 1999. Light grey, crop and pasture land; medium grey, shrubland and arboreous shrubland; black, native forest.

Trends in forest fragmentation

The analyses of landscape indices revealed that the loss of native forests of the four study areas has been associated with substantial forest fragmentation. Spatial patterns of forest loss and fragmentation of each study area are reported below.

Central Veracruz, Mexico
Mean size of forest fragments increased slightly from 1176 ha in 1984 to 1291 ha in 2000 (Table 2.3). However, the size of the largest fragment decreased by more than a half during the study period from 75,105 ha to 30,980 ha. Patch density did not show a substantial variation over time. The largest patch index decreased from 5.5% in 1984 to 2.2% in 2000. Similarly, the total edge length and total core area presented a decline during the study period as a result of the fragmentation of remnant forest. These changes in the spatial configuration of the landscape were associated with a decline in mean proximity of more than 60%, as result of the division of forest fragments.

The Highlands of Chiapas, Mexico
During the first time interval, the mean patch size decreased by approximately 58%. Between 1990 and 2000, this index exhibited a greater decline of 67%. Patch density increased overall through time, reaching its maximum value of 3.2 patches per 100 ha in 2000. This pattern was associated with a rapid decline in the largest patch index, from 60.7% in 1975 to 35.1% in 1990, and to 4% in 2000. From 1975 to 1990, the total edge length increased by approximately 100%, as a result of the progressive fragmentation. However, for the second time interval, this index experienced a decline of 23%, owing to the loss of forest fragments. Approximately 70% of the total core area recorded in 1975 had disappeared by 1990. Similarly, a high percentage of decline in total core area (67%) was observed between 1990 and 2000. The process of fragmentation was also accompanied by an increase in fragment isolation, indicated by a rapid decline in the mean proximity over time. Between 1975 and 2000, this index decreased by approximately 98%.

Rio Maule-Cobquecura, Chile
During the first study period, the native forests were mainly affected by severe fragmentation (increasing number of patches) and deforestation (decreasing mean patch size). For the second time interval, deforestation became the dominant process, owing to a decline in both mean patch size and patch density. This trend was associated with a reduction in the size of the largest forest patch, ranging from 7% of the total area in 1975 to 0.2% in 2000. The landscape was also characterized by the presence of more patch edges, which indicates that the shape of native forest patches had become more irregular during the first time interval. However, between 1990 and 2000 the total edge length in the landscape declined as a result of the loss of forest fragments. The native forest fragments showed a substantial decrease in the total amount of core area and in the mean proximity over time. Between 1975 and 1990, the total core area decreased from 21,138 ha to 918 ha, and then to 839 ha in 2000. Similarly, the main change in the mean proximity was recorded in the first time interval. During this period, the neighbourhood of forest patches rapidly became occupied by areas of a different land-cover type, as native forest patches became spatially separated and less contiguous in distribution.

Table 2.3. Changes in landscape pattern indices of the native forests in each study area. Numbers in parentheses correspond to minimum and maximum values.

Central Veracruz, Mexico

Landscape indices	1984	2000
Mean patch size (ha)	1,176 (1–75,105)	1,291 (1–30,980)
Patch density (per 100 ha)	0.013	0.009
Largest patch index (%)	5.5	2.2
Total edge length (km)	5,276	4,222
Total core area (ha)	177,031	131,020
Mean proximity	860 (0.0–37,555)	326 (0.0– 8,951)

The Highlands of Chiapas, Mexico

Landscape indices	1975	1990	2000
Mean patch size (ha)	65.0 (0.5–211,180)	26.9 (0.5–119,516)	8.67 (0.5–13,279)
Patch density (per 100 ha)	1.0	1.9	3.2
Largest patch index (%)	60.7	35.1	4.0
Total edge length (km)	24,781	50,114	38,400
Total core area (ha)	99,422	29,860	9,611
Mean proximity	101,369 (0.02–587,150)	60,017 (0.0 –342,240)	1,405 (0.0 –34,466)

Rio Maule-Cobquecura, Chile

Landscape indices	1975	1990	2000
Mean patch size (ha)	17 (0.5–52,178)	5 (0.5–9,842)	4 (0.5–1,182)
Patch density (per 100 ha)	0.93	1.65	1.36
Largest patch index (%)	6.91	1.30	0.16
Total edge length (km)	20,330	22,337	15,799
Total core area (ha)	21,138	918	839
Mean proximity	5,880 (0.0–145,119.4)	612 (0.0–29,276.4)	73 (0.0–6,031.6)

Los Muermos-Ancud, Chile

Landscape indices	1976	1985	1999
Mean patch size (ha)	47 (0.5–132,971)	24 (0.5–49,767)	19 (0.5–42,785)
Patch density (per 100 ha)	0.36	0.60	0.65
Largest patch index (%)	8.31	3.11	2.67
Total edge length (km)	21,403	30,931	31,072
Total core area (ha)	143,428	89,007	69,900
Mean proximity	19,350 (0.0–369,603.5)	4,380 (0.0–152,583.1)	2,552 (0.0–120,135)

Los Muermos-Ancud, Chile

The mean size of forest patches decreased gradually from 47 ha in 1976 to 24 ha in 1985 to 19 ha in 1999 (Table 2.3). This decline in the patch size was associated with a continuous increase in the patch density over time, reaching its maximum value of 0.65 fragments per 100 ha in 1999. This pattern was accompanied by a reduction of the largest forest patch, from 8% in 1976 to 3% of the total area in 1999. This modification of the landscape was also characterized by the presence of more forest patch edges, which increased during the first study period. In the second time interval, the total edge length showed a slight increase. The total core area also showed a gradual decline across the time intervals, by 1999 decreasing by more than 50% of the core area recorded in 1976. The main change in fragment isolation occurred from 1976 to 1985, when mean proximity decreased to almost one-fifth of its initial value. Between 1985 and 1999, the mean proximity presented a further decline.

Drivers of deforestation

Multiple logistic regression models indicated which explanatory variables were significantly related to the probability of deforestation in each study area (Table 2.4). In Central Veracruz the logistic regression model revealed that the probability of an area being cleared of forest for the 1984–2000 interval was highly significant and negatively related to slope and distance to patch edges (Table 2.4). For the same study period, soil fertility was highly positively related to deforested areas. Distance to towns and population density in 1995 were negatively associated with the probability of deforestation, whereas distance to national parks, mean annual rainfall and distance to agricultural areas appeared to be positively related.

In the Highlands of Chiapas the logistic regression model showed that slope and distance to patch edges were significantly associated with deforested areas for the 1975–1990 interval. Conversely, density of non-forest areas and patch size appeared to be positively related to the probability of deforestation. For the 1990–2000 interval, elevation, distance to patch edge and slope were statistically negatively related to deforested areas. Conversely, patch size was positively associated with the probability of deforestation.

In Rio Maule-Cobquecura, the probability of an area being cleared of forest for the 1975–1990 interval was negatively related to distance to patch edge and patch size. For the 1990–2000 interval distance to patch edge and slope were negatively associated with deforested areas. In Los Muermos-Ancud, slope and distance to patch edge were statistically negatively related to the loss of native forests during the first time interval. In the second time interval, distance to patch edge, patch size and slope appeared to be highly negatively associated with the clearance of forested areas. The regression model also revealed that distance to rivers was positively related to the probability of deforestation.

Table 2.4. List of coefficients for variables significantly affecting the probability of deforestation in Mexico and Chile. Results were obtained from multivariate logistic regression modelling. The model is logit (P) = X, where P is the probability of deforestation and X is a linear combination of explanatory variables.

Variable	Order	Coefficients	Std Error	χ^2	P-value
Central Veracruz, Mexico					
Intercept	1	−0.8771	0.59770	2.1	n.s.
Slope	2	−0.0628	0.00943	44.3	***
Distance to national parks	3	0.0001	0.00001	5.2	*
Mean annual rainfall	5	0.0007	0.00023	9.9	**
Population density in 1995	6	−0.0015	0.00051	8.9	**
Soil fertility	7	0.5664	0.16270	12.1	***
Distance to towns	8	−0.0003	0.00011	7.0	**
Distance to agricultural areas	9	0.0002	0.00009	5.3	*
Distance to patch edge	4	−0.0010	0.00024	17.8	***
Estimated by stepwise logistic regression procedure of SAS. Order of entry into the model is provided. Df = 1.					
The Highlands of Chiapas, Mexico					
Period 1: 1975–1990					
Intercept		$-8.045\ 10^{-1}$	$3.848\ 10^{-1}$	2.091	**
Slope		$-4.577\ 10^{-2}$	$7.770\ 10^{-3}$	5.891	***
Density of non-forest areas		2.112	$4.908\ 10^{-1}$	4.303	***
Distance to patch edge		$-4.869\ 10^{-3}$	$9.277\ 10^{-4}$	5.249	**
Patch size		$3.648\ 10^{-6}$	$1.318\ 10^{-6}$	2.769	***
Period 2: 1990–2000					
Null model					
Intercept		4.034	$5.479\ 10^{-1}$	7.363	***
Elevation		$-1.759\ 10^{-3}$	$2.311\ 10^{-4}$	7.613	***
Distance to patch edge		$-3.087\ 10^{-3}$	$5.740\ 10^{-4}$	5.377	***
Patch size		$4.104\ 10^{-6}$	$1.557\ 10^{-6}$	2.635	***
Slope		$-1.839\ 10^{-2}$	$7.243\ 10^{-3}$	2.540	**
N = 1242 points (1975–1990) and 992 points (1990–2000). Df = 1.					
Rio Maule-Cobquecura, Chile					
Period 1: 1975–1990					
Intercept		1.424	$1.060\ 10^{-1}$	13.436	***
Distance to patch edge		$-4.891\ 10^{-3}$	$1.121\ 10^{-3}$	−4.365	***
Patch size		$-9.009\ 10^{-6}$	$3.186\ 10^{-6}$	−2.827	**
Period 2: 1990–2000					
Null model					
Distance to patch edge		−0.017	0.004	−3.711	***
Slope		−0.032	0.008	−3.912	***
N = 1489 points (1975–1990) and 622 (1990–2000). Df = 1.					
Los Muermos-Ancud, Chile					
Period 1: 1976–1985					
Intercept		0.065	0.113	0.573	n.s.
Slope		−0.049	0.014	−3.371	***
Distance to patch edge		−0.006	0.001	−8.012	***

Continued

Table 2.4. *Continued*

Variable	Order	Coefficients	Std Error	χ^2	*P*-value
Period 2: 1985–1999					
Intercept		$1.102\ 10^{-1}$	$1.088\ 10^{-1}$	1.013	n.s.
Distance to rivers		$2.760\ 10^{-4}$	$1.152\ 10^{-4}$	2.395	*
Distance to patch edge		$-3.341\ 10^{-3}$	$8.134\ 10^{-4}$	−4.108	***
Patch size		$-1.762\ 10^{-5}$	$4.000\ 10^{-6}$	−4.406	***
Slope		$-5.716\ 10^{-2}$	$1.494\ 10^{-2}$	−3.826	***
N = 1000 points in both periods. Df = 1.					

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$; n.s., not significant.

Discussion

Forest loss and land-cover change in Mexico and Chile

The native forests of the four study areas have undergone relatively high rates of forest loss during the decades analysed, compared to many other forested landscapes in the world (Spies *et al.*, 1994; Zheng *et al.*, 1997; Cushman and Wallin, 2000; Cohen *et al.*, 2002; Staus *et al.*, 2002). These forests have been reduced severely and degraded over time owing to logging for timber and fuelwood, and clearance for cultivation. Of all the study areas, Rio Maule-Cobquecura had the highest rate of deforestation in the last three decades (4.5% per year), followed by the Highlands of Chiapas (3.4% per year). Central Veracruz and Los Muermos-Ancud presented lower rates of 2% and 1.1% per year respectively. By analysing the rate of forest loss by time intervals, in the 1990–2000 interval the Highlands of Chiapas had the highest rate of deforestation (6.2% per year). In the two Chilean study areas, the highest rates of forest loss were recorded during the first time interval. As for the total reduction of native forests, Rio Maule-Cobquecura and the Highlands of Chiapas showed the highest losses of 67% and 57% in the last three decades, respectively. Central Veracruz and Los Muermos-Ancud exhibited lower rates of 26% for the 1984–2000 interval and 23% for the 1976–1999 interval, respectively.

Forest losses recorded in other studies, assuming that they were calculated using the FAO formula, have generally been lower. A forest loss rate of 0.5% per year was estimated for the Klamath-Siskiyou ecoregion, USA, and an overall (cumulative) reduction of forest cover by 10.5% was recorded over the period 1972–1992 (Staus *et al.*, 2002). In western Oregon, deforestation rates by clearcutting between 1972 and 1995 varied from 0.5% to 1.2% per year with almost 20% of the total forest impacted (Cohen *et al.*, 2002). Similarly, in other areas of western Oregon, between 1972 and 1988 the rate of deforestation, primarily by clearcutting, was 1.2% per year of the entire study area including the wilderness area (Spies *et al.*, 1994). A rate of 0.6% per year, slightly lower than that determined for the period 1990–2000 in Los Muermos-Ancud, was found for the 1986–1996 interval in the Napo region of western Amazonia (Sierra, 2000). A rate of 6% per year was determined for lowland deciduous forest in eastern Santa Cruz, Bolivia in the middle

1990s (Steininger *et al.*, 2001), thought to be one of the highest deforestation rates reported anywhere in the world. However, the present study reported a slightly higher rate for the 1990–2000 interval in the Highlands of Chiapas.

All of the study landscapes were affected by substantial changes in the area of the different land-cover types over time. The greatest losses of native forests in the study areas were associated with the conversion to human-induced land-cover types. In the Highlands of Chiapas and Los Muermos-Ancud the loss of native forests has been related to an increase in the area of crop and pasture lands. Conversely, in Central Veracruz there was an expansion of forest cover owing to the abandonment of crops. The loss of native forests was mainly determined by the degradation of pristine forest into old-fields (including disturbed secondary forests with herbaceous and shrub cover), although transformation for crops and cattle ranching also played an important but lesser role. In Rio Maule-Cobquecura, a substantial area of native forests was converted to exotic-species plantations such as *Pinus radiata* and *Eucalyptus* spp.

Spatial patterns of forest loss and fragmentation in Mexico and Chile

Landscape pattern indices provide a useful tool to explore cross-site differences and changes over time. The simultaneous use of class-level and patch-level landscape pattern indices enabled assessment of the spatial configuration of forest cover and its relation to principal land-cover types. It is important to highlight that, owing to the different spatial scale of the data on which these analyses were performed (1:250,000 scale) for Veracruz, the analysis of fragmentation generated some differences compared to the other study areas.

Forest fragmentation has three recognizable components at the landscape level: (i) habitat loss; (ii) reduction of patch size; and (iii) increased isolation of habitats (Bennett, 2003). These three components were shown to occur over the last decades in the four study areas analysed in Mexico and Chile. In particular, the mean size of forest fragments declined consistently over time, except in Central Veracruz, which displayed a similar pattern to the situation recorded in Wisconsin (Pan *et al.*, 1999), where the size of fragments increased due to abandonment of agricultural land. Over the last three decades, the greatest reduction in the mean patch size was recorded in the Highlands of Chiapas, followed by Rio Maule-Cobquecura. However, Rio Maule-Cobquecura reached the smallest size of forest fragments in the last study interval. These results support the statement made by Armenteras *et al.* (2003) that progressive reduction in the size of forest habitats is a key component of ecosystem fragmentation.

Patch density increased gradually in the Highlands of Chiapas and Los Muermos-Ancud, except in Central Veracruz. In Rio Maule-Cobquecura patch density reached its maximum value in 1990 and then decreased by 2000. A similar trend was observed in the total edge length in the Highlands of Chiapas and Rio Maule-Cobquecura, which increased until 1990 and then declined by 2000. This pattern reflects an increase of patch density and edge length in the earliest stages of forest loss and fragmentation and a decline during the later stages of deforestation. Zipperer *et al.* (1990) also observed that the constant action of deforestation led to a decline in patch density in central New York, USA. In Rio Maule-Cobquecura, this process even eliminated forest patches created during the first study period. Similarly to the findings of Ranta *et al.* (1998), the substantial increase of patch density in Rio Maule-Cobquecura was related to the concentration of the forest area in patches less than 100 ha in area.

As recorded elsewhere (Fitzsimmons, 2003), the greatest absolute decline in the largest forest patch size in the Highlands of Chiapas and Los Muermos-Ancud coincided with the time period where the greatest absolute decline in annual forest loss was observed. In Rio Maule-Cobquecura, a slightly higher decline was observed in the time interval that did not present the highest rate of deforestation. The decline of large forest fragments might have a significant effect on the response of some species in the study area. For instance, the size of the largest cloud forest fragments was the most important characteristic influencing the response of bird species in eastern Mexico (Martínez-Morales, 2005). Similarly, higher bird species richness of resident and migrant species occurred in larger forest fragments in Singapore Island (Castelletta *et al.*, 2005).

Interior forest habitat decreased progressively over time in all of the study areas. Also, forest fragments became more isolated as other land-cover types occupied the deforested areas in the study landscapes. Rio Maule-Cobquecura and the Highlands of Chiapas were characterized by substantial reductions in the total core area (96% and 90%, respectively) over the last three decades, while Los Muermos-Ancud and Central Veracruz presented lower reductions (51% and 26%, respectively). Reductions in the mean proximity over the study periods were also higher in Rio Maule-Cobquecura and the Highlands of Chiapas, with 98.7% and 98.6%, respectively. In Los Muermos-Ancud and Central Veracruz, this index declined to a lower percentage of 86.8% and 62%, respectively.

The analysis of spatial patterns of landscape indices needs to be understood as a first step to comprehend ecological processes, and not as an end itself (Li and Wu, 2004). Although these indices allow forest fragmentation to be assessed at the landscape level, it is necessary to explore the relationships between pattern and process. A variety of studies that relate spatial patterns to ecological processes have demonstrated that forest fragmentation may lead to a change in the abundance and richness of some woody (Metzger, 1997, 2000) and bird species (Willson *et al.*, 1994; Cornelius *et al.*, 2000; Drinnan, 2005; Martínez-Morales, 2005; Uezu *et al.*, 2005). Therefore, the loss of forest habitats and the increasing trend of fragmentation over forthcoming decades in the study areas may have negative consequences on the flora and fauna existing in the remnant forests, due to changes in composition of assemblages and changes in ecological processes (Forman and Godron, 1986; Bennett, 2003) (see Chapter 3).

Causal factors of deforestation

The driving factors of deforestation identified by the spatially explicit models are all variables that express geophysical attributes or the 'symptoms' of the underlying causes of forest loss. These factors show how the forest loss has taken place spatially and temporally in the landscapes, but they do not necessarily show the underlying causes. In fact, these factors are the result of cultural and socio-economic processes that have been modifying the landscape over many decades. Bürgi *et al.* (2004) define these factors as 'attractors of change', as they are the primary driving forces likely to induce change at a local scale.

According to the models generated for the different time intervals, deforestation was essentially concentrated in gently sloping areas as a result of the expansion of crop and pasture lands in Mexico and Chile, except for the first time interval in Rio Maule-Cobquecura. In this landscape the process of deforestation was not significantly related to slope owing to the conversion of native forests to exotic plantations in sites of different degrees of slope between 1975 and 1990. Similarly to results obtained for the second time interval in Rio Maule-Cobquecura and for all time intervals in Los Muermos-Ancud and the two study areas in Mexico, Wilson *et al.* (2005) found that slope is a highly significant variable for explaining the probability of deforestation. In particular, these authors found that forested flat areas near towns and roads were highly vulnerable to the conversion of native forest to industrial plantations of exotic species. In contrast to that study, the logistic regression of the present work revealed that distance to towns and distance to roads were not significant in accounting for clearance of forest area, except in Central Veracruz, where the distance to towns was a significant factor influencing deforestation.

Results also revealed that the clearance of forests was concentrated around edges of forest fragments in all of the study areas. A logistic model-based study conducted in Madagascar similarly found that the expansion of agriculture into the remaining natural forest was associated with progressive clearance from forest edges (McConnell *et al.*, 2004).

In most of the study areas small patches became vulnerable to deforestation, owing to the fact that they were mainly concentrated in flat areas, where the process of deforestation was more intense. In particular, the severe fragmentation reported for both time intervals in the Highlands of Chiapas and for the first time interval in Rio Maule-Cobquecura led to an increase in the abundance of smaller patches that were subsequently eliminated by expansion of agriculture and plantations of exotic species respectively. In Los Muermos-Ancud the loss of forests between 1985 and 1999 was concentrated in small fragments located away from rivers or streams. The significance of this driver is related to the fact that clearance of forests is legally prohibited in areas close to rivers. This prohibition was more evident in flat areas where the forest patches were intensely fragmented and left as riparian vegetation. Similarly, the application of logistic regression to analyse the decline of native grassland in Melbourne, Australia revealed that patches close to streams were associated with a low probability of being destroyed (Williams *et al.*, 2005). In the Highlands of Chiapas the loss of native forest was located in lowlands owing to the presence of steep slopes in the highlands.

These cultural and socio-economic factors do not by themselves describe the immediate causes of forest cover change, but are related to various environment policies. In fact, some assessments indicate that neither population growth nor poverty alone constitutes the sole and major underlying cause of land-cover change worldwide (Lambin *et al.*, 2001). Rather, people's responses to economic opportunities, mediated by institutional factors, drive land-cover changes. For instance, population and income variables were found to be significant factors explaining forest area variation between 1970 and 1991 in 67 tropical countries (Uusivuori *et al.*, 2002). However, these results do not explain directly the causes of deforestation, as they need to be linked with international forest policies in order to understand the deforestation at the regional level. At a local level, people's response to institutional support has been documented in Bangladesh, where farmers abandoned extensive shifting cultivation, adapting suitable commercial land uses such as agroforestry, horticulture and forest plantations (Rasul *et al.*, 2004). Conversely, in Chile the context of a strong free market economy, dominated by economically powerful private domestic and international pulp and paper companies, has led to a market-friendly forest policy (Silva, 2004). As a result of this economy, many principles of sustainable development have been violated, causing a series of negative impacts on the environment (Lara and Veblen, 1993; Lara *et al.*, 2000).

Compared to Chile, Mexican industrial timber interests were relatively weak over the interval studied, and the forest peasant sector was much stronger and better organized, with the result that timber interests could not dominate forest communities as they could in Chile (Silva, 2004). The *ejido* land tenure system provided a platform for organizing political and economic activity that was not available in Chile. However, large-scale timber interests gained significant support in Mexico in the 1990s owing to the support of the presidency (Silva, 2004). The increase in the rate of deforestation during the 1990s in the Highlands of Chiapas reflects the effect of changes in social and economical policies in this country over the last decade. For instance, the lack of governance following the Zapatista rebellion in 1994, which allowed rampant illegal clearing for agriculture, livestock ranching and human settlement (González-Espinosa, 2005), did not help forest conservation. The contrasting cases of Chile and Mexico provide significant insight into the conditions needed for an improvement in national forest policies. Although the policy environment and socio-economic circumstances are very different in the two countries, the end result – high rates of deforestation and forest fragmentation – has been the same.

Future trends of deforestation

In the last three decades, substantial changes in the land-cover types and in the spatial configuration of native forests were recorded across the study areas. Expansion of human-induced land-cover types such as crops and pasture lands, forest plantations of exotic species and degraded secondary forests were associated with a considerable loss of native forest in each study area. With progressive forest loss and fragmentation, the native forests presented abrupt changes in their spatial configuration over the whole study period, from a forest habitat formed by complex clusters of large fragments to a sparse distribution of smaller patches.

Based on the current trends of deforestation, and if the primary driving forces of deforestation continue operating, we expect a continuous loss and fragmentation of native forests during forthcoming decades in Mexico and Chile. The forest area of patches corresponding to the smallest size class will tend to decline as a result of the clearance of small fragments that still exist in gently sloping sites. Furthermore, a decline in patch density might be observed in coming decades in Central Veracruz, the Highlands of Chiapas and Los Muermos-Ancud. This decline in the curve of patch density was recorded in the 1990s in Rio Maule-Cobquecura, reflecting the endpoint of the deforestation process: a landscape largely devoid of natural forest.

Conclusions

This study has succeeded in characterizing the major changes in forest configuration that have taken place over the past three decades in Mexico and Chile. The land-cover change analysis demonstrated that the landscapes are becoming increasing dominated by crops and pasture and by forest plantations of exotic species. Results also demonstrated that the patterns of deforestation have had a notable effect on the spatial configuration of the remaining forest fragments. As a result, the study landscapes have become dominated by isolated, small forest fragments. This pattern exposes how native forests are being disturbed spatially, which in turn illustrates the effects of socio-economic drivers of deforestation, such as forest logging and clearance for crops and pasture land. These causes are dependent on underlying social and economic policies, which in reality drive land-cover change. The assessment of these local causal relationships can potentially inform the development of improved land-use policies and management. However, the cases of Mexico and Chile provide evidence of similar patterns of forest loss and fragmentation in four different landscapes affected by human activities, despite contrasting policy environments and socio-economic characteristics.

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