

## 2 ASSESSING THE CURRENT EXTENT AND RECENT LOSS OF DRYLAND FOREST ECOSYSTEMS

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*J.M. Rey Benayas, L. Cristóbal, T. Kitzberger, R. Manson, F. López-Barrera, J. Schulz, R. Vaca, L. Cayuela, R. Rivera, L. Malizia, D. Golicher, C. Echeverría, R. del Castillo, J. Salas*

### Introduction

Land-cover change is regarded as the most important global change affecting ecological systems (Vitousek, 1994). Natural landscapes – i.e. those largely unaffected or hardly affected by human activities – are being rapidly transformed into urban and farmland landscapes throughout the world (Foley *et al.*, 2005; Feranec *et al.*, 2010; López and Sierra, 2010). As the characteristics of land-cover have important impacts on climate, biogeochemistry, hydrology, species diversity, and the well-being of human societies, land-cover change has been identified as a high priority for research and to inform the development of strategies for sustainable management (Turner *et al.*, 1993; Ojima *et al.*, 1994; Millennium Ecosystem Assessment, 2005a). In recent years, special attention has been given to land-use changes and degradation in drylands. Dryland forests are highly prone to degradation and desertification on account of their limited primary productivity and slow recovery following human disturbance (Millennium Ecosystem Assessment, 2005b), yet these ecosystems play a crucial role in providing services such as climate and water regulation (Maass *et al.*, 2005; Lemons, 2006).

To develop approaches for the conservation and restoration of dryland forests at the regional level it is crucial to know their current extent and to understand the main recent and historical changes that have affected them (Schulz *et al.*, 2010). To accomplish this goal, it is necessary to assess what processes may be driving such changes, to reveal the threats to forest ecosystems, and to develop alternative strategies to diminish these threats (Angelsen and Kaimowitz, 1999; Geist and Lambin, 2002; Lambin *et al.*, 2003; Antrop, 2005; Olander *et al.*, 2008). Changes in patterns of forest distribution at a wide variety of spatial scales, from global to local scales, are among the land-cover changes most frequently investigated. At the global scale, forest extent is declining mostly as a result of expansion of farmland (Foley, 2005; FAO, 2010). Additional activities associated with widespread deforestation and forest degradation include industrial tree plantations (often composed of non-native species), logging for firewood, and cattle grazing (Lara and Veblen, 1993; Kahn and McDonald, 1997; Geist and Lambin, 2002). However, at local and regional scales, land abandonment resulting mainly from rural-urban migration can contribute towards passively restoring considerable amounts of the original forest extent (Aide and Grau, 2004; Pascarella *et al.*, 2000; Rudel *et al.*, 2005; Grau and Aide, 2008; Parés-Ramos *et al.*, 2008). These landscape processes have rarely been mapped and quantified, and change trajectories among land-cover types have not been systematically examined for particular types of forest.

We addressed these issues in selected areas in Chile, Argentina and Mexico, using standardized research protocols (Box 2.1). The primary advantage of assessing these

different areas is that they include a range of ecological, socioeconomic, and cultural characteristics. We used remote sensing data to measure and monitor land-cover change because of their ability to capture an instantaneous synoptic view of a large part of the Earth's surface and to provide repeated measurements of the same area on a regular basis (Donoghue, 2002). Land-cover change detection and monitoring is especially useful in those regions where there is a lack of available cartographic information with sufficient spatial resolution to examine land-cover change. To investigate the possible causes of change in forest cover, a Geographic Information System (GIS) database incorporating satellite imagery and biophysical and socioeconomic variables was developed for each study area. This information was statistically analyzed to infer likely drivers of forest-cover change and to test a series of specific hypotheses relating to the factors responsible for deforestation. For example, we expected that the rate of forest loss would be (i) positively associated with population density and accessibility (i.e. proximity to roads and rivers), and (ii) highest on sites most suitable for agriculture such as those with gentle slopes (Sader and Joyce, 1988; Pfaff, 1999; Lambin *et al.*, 2003).

As part of the contribution to the international ReForLan project (Newton, 2008), we report here the first multi-regional assessment of land-use/land-cover changes, with special attention to forest loss, spanning a ca. 30-year period (1970s–2000s) in dryland Latin American regions. The results presented should be useful for planning the restoration and conservation of dryland forest in the study areas and elsewhere in the region. Furthermore, they provide an example of the research that needs to be conducted in other regions of the world.

### **Box 2.1** Methodology used to assess the amount and the drivers of forest change

A total of six areas were considered in this study in central Chile (the Central Valley and the Coastal Range extending to the Pacific Ocean), southern Argentina (northwestern Patagonia), northern Argentina (Salta province), as well as central Veracruz (central Mexico by the Gulf Coast), Central Depression of Chiapas, and Oaxaca (southern Mexico by the Pacific Coast).

#### **Analysis of land-cover/land-use change**

##### *Remote sensing data*

We acquired a time-series of satellite imagery to analyze land-cover/land-use change in each study area (**Table 1**). All images were pre-processed, including geometric, atmospheric and topographic corrections. The images were geometrically corrected using standard procedures based on ground and roadway map control points. For the removal of atmospheric effects and variations in solar irradiance, an atmospheric correction was carried out to transform the original radiance images to reflectance images using an algorithm based on the Chavez reflectivity model (Chávez, 1996) in most study areas. Topographic corrections were also performed to reduce shadows on hilly areas when necessary. We employed a variety of methods for this task, such as the C-correction proposed by Teillet *et al.*, (1982) using a digital elevation model (DEM) interpolated from contour lines of 25 m for TM and ETM+ images in central Chile and Oaxaca, and the NASA SRTM model with a resolution of three arc seconds per pixel (ca. 80 m) in central Veracruz. To compare images of different pixel size, the original MSS raster grids were re-sampled to the resolution of the TM raster grids (30 m) in most study areas.

**Box 2.1 (cont.)****Table 1** Time series of satellite imagery (Landsat and SPOT) used to analyze land-cover/land-use change in the selected study areas in Latin America.

Study area	Year/Sensor	Year/Sensor	Year/Sensor	Year/Sensor
Central Chile	1975 MSS	1985 TM	1999 TM	2008 ETM+
Southern Argentina	1973 MSS	1985 TM	1997 TM	2003 ETM+
Northern Argentina	1976/77 MSS	1987 TM	1993 TM	2006 TM
Central Veracruz (Mx)	1973 MSS	1990 TM	2000 ETM+	2007/08 SPOT
Chiapas (Mx)	—*	1990 TM	2000 ETM+	2005 ETM+
Oaxaca (Mx)	1979 MSS	1989 TM	2000 ETM+	2005 SPOT

\*Classification results for 1975 were excluded due to inconsistencies related to Landsat MSS resolution.

*Land-cover/land-use classification*

All study areas attempted to follow a common protocol for classification of land-cover/land-use types that distinguished eight major classes: (1) forest, (2) shrubland, (3) pasture, (4) bare ground, (5) agriculture, (6) timber plantations, (7) urban areas, and (8) water. However, these pre-defined classes were modified according to local conditions in each study area. The land-cover maps were derived using a supervised classification procedure in all study areas except in northern Argentina, where an ISODATA unsupervised classification technique was used.

To classify the images, field points were taken with a GPS in order to train the spectral signature of the selected land-cover classes (198 in central Chile, 311 in southern Argentina, 1071 in central Veracruz, and 50 in Oaxaca). This information was complemented with high-resolution imagery obtained from Google Earth and aerial ortho-photos to account for areas with restricted accessibility, control points from vegetation and land-use maps. In addition, informal interviews were conducted with land owners and land managers during field surveys to obtain information on previous and current land-cover and land use.

For central Chile and Oaxaca, a region-growing approach was used with the 'seed' function, and signature separability of the initial classes for all images was evaluated using the Bhattacharyya distance. Based on this distance, classes were iteratively merged until reasonably high signature separability (Bhattacharyya distance >1.9) was achieved. For northern Argentina, the classifications performed used nine iterations and a convergence of 0.95. For central Veracruz, polygons were created by means of a regional growth algorithm that combined pixels that were judged to be similar based on the values of the spectral bands available for each image, as well as data from scale, texture, and shape indices. For central Chiapas, classifications were performed using an iterative procedure developed for classifying multi-temporal Landsat imagery in complex tropical landscapes (Harper *et al.*, 2007). A unified classification for the three analyzed years was produced. Maximum likelihood classification was the principal method used as it has proven to be a robust and consistent classifier for multi-date classifications (Yuan *et al.*, 2005); however, the Sequential Maximum A-Posteriori classifier (SMAP in GRASS 6.4) produced the best results in Chiapas. Post-classification processing was applied to combine initial classes and to better discriminate between confounding classes in each study area. More details on the classification procedures and used software can be found in Rey Benayas *et al.*, (2010a), Schulz *et al.*, (2010), Cristóbal *et al.*, (in prep.), Gowda *et al.*, (in prep.), Manson *et al.*, (in prep.), Rivera *et al.*, (in prep.), and Vaca *et al.*, (in prep.).

## **Box 2.1 (cont.)**

### *Accuracy assessment*

Accuracy assessment of the classification results was carried out using independent ground control points sampled in the field (280 in central Chile, 432–520 control points in southern Argentina, 157 in northern Argentina, and 300 in Oaxaca) as well as control points derived from Google Earth and ortho-photos (>400 in central Veracruz, and >2500 in Chiapas). Based on these points, confusion matrices and associated Kappa indexes of agreement for each class were generated (Rosenfield and Fitzpatrick-Lins, 1986). As independent data sources such as previous classifications and ortho-photos were largely absent for central Veracruz, an Index of Ecological Congruence (IEC) was developed following the same logic as the Kappa index, where congruent and incongruent events were considered as feasible or non-feasible land-cover transitions between two time periods given current ecological understanding of land-use patterns and rates of vegetation succession in the region. Overall classification accuracies for the different classified study areas are reported in **Table 2**.

**Table 2** Classification accuracy (%) in the different study areas and images. The sequence of images (columns) follows **Table 1**.

Study area	Series 1	Series 2	Series 3	Series 4
Central Chile	68.5	77.3	78.9	89.8
Southern Argentina	69.3	80.7	85.0	86.0
Northern Argentina	68.2	82.5	85.3	83.4
Central Veracruz (Mx)*	93.4	94.0	74.8	82.0
Chiapas (Mx)	—	74.1**		
Oaxaca (Mx)	92	90	93.3	93.5

\*Based on an Index of Ecological Congruency (IEC) and not Kappa for Series 1–3 (1973–2000). \*\*Since a unified classification for the three analyzed years was produced, we performed a unique validation analysis.

### *Change identification*

The spatial distribution of land-cover/land-use changes was investigated using the previously classified remotely sensed images in order to obtain a matrix of change directions among land-cover classes (Lu *et al.*, 2004) in each study area. Changes were analyzed by cross-tabulation as proposed by Pontius *et al.*, (2004) to quantify net changes, gains, losses and persistence as well as inter-categorical change trajectories. In Chiapas, the classification was combined with results from the map series IV (2007–9) of Use of the land and Vegetation elaborated by INEGI (scale 1:250,000).

## **Drivers of forest cover change**

### *GIS analysis and explanatory variables*

To analyze the drivers of forest cover change we created binary maps to represent forest versus other cover types. The points where change (deforestation) occurred or not were determined by overlapping the binary land-cover maps from the two dates that spanned the time period studied in each study area. In southern Argentina, since afforestation and not deforestation was detected, we analyzed drivers of afforestation with exotic conifers rather than drivers of deforestation.

## **Box 2.1 (cont.)**

We then followed a common protocol (Echeverría *et al.*, 2006) that included randomly selecting a grid of sampling points separated by a minimum distance of 1000 m for extracting values of both the response (change vs. no change) and explanatory variables (biophysical and socioeconomic) in order to reduce problems with spatial auto-correlation. A number of biophysical and socioeconomic variables that may influence forest cover change were selected for analysis including: (1) elevation (m), (2) slope (°), (3) insolation or radiation input, (4) mean annual precipitation (mm), (5) soil quality, (6) distance from rivers and lakes (m), (7) distance from forest edges (only for forest loss), (8) distance from forest (only for forest gain in southern Argentina), (9) distance from human settlements (cities, towns and villages) with different numbers of inhabitants (m), (10) human population density (#/km<sup>2</sup>), (11) distance from different types of roads (e.g. primary or secondary, paved or unpaved (m), (12) distance from the agricultural frontier (m), (13) distance from irrigation infrastructure (m), and (14) distance from cattle pastures (m). For all explanatory variables, values were extracted using the random sampling points previously selected with forest to non-forest changes for each of the four analysis periods. However, not all of these variables were used in each study area as availability of information depended on local conditions.

### *Model building*

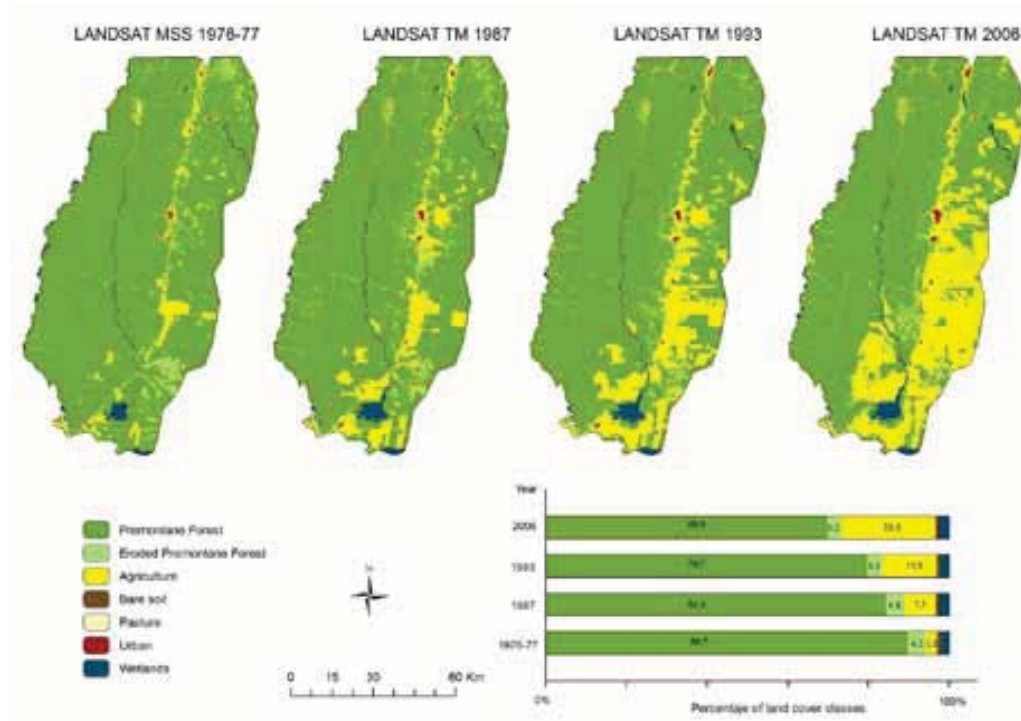
Logistic regressions were performed to explore the effect of the described explanatory variables on deforestation in all study areas except in Chiapas, where a different method was used (see below). To fit the models, we started by using the full set of explanatory variables described above. Before starting the model selection process, a Pearson's or Spearman's correlation test was performed to identify the correlated explanatory variables, and the single representative of variables that were highly correlated (typically  $r > 0.7$ ) was selected for further analyses to avoid multicollinearity. A spatial correlogram based on Moran's index of autocorrelation was used to explore the autocorrelation of data at different geographic distances, and was found to be low in all cases. In southern Argentina, the effect of each predictor variable on the occurrence of afforestation was evaluated by univariate logistic models. In Chiapas, a series of generalized additive models (GAMs) of the binomial family were fitted using the R package 'MGCV' (Wood, 2004). These models allow non-linear responses to be modelled.

### *Model selection*

Multivariate, spatially explicit models were developed for each of the three periods of time and the whole study period in most study areas. In central Chile and northern Argentina, we performed a backward stepwise model selection based on the Akaike (1974) Information Criterion (AIC) to determine the set of explanatory variables constituting the best fitting model for each period. The measure used by generalized linear models, including logistic regression, to assess the aptness of fit is called the deviance. Deviance reduction ( $D^2$ ) is estimated as  $D^2 = (\text{Null deviance} - \text{Residual deviance}) / \text{Null deviance}$ . For southern Argentina, selection of the final multivariate logistic model to explain afforestation was performed using forward, backward and best subset procedures with GIS Landchange Modeler (Idrisi, 2006); then a spatial model of potential afforestation was generated and evaluated by means of the relative operating characteristic (ROC) curve. For central Veracruz, those variables with  $p \leq 0.10$  in univariate logistic regressions were used to construct a final multiple logistic regression model that best explained the loss of undisturbed forest in the study region. In Chiapas, to complement the GAM analysis and provide a direct interpretation of the strength of the drivers, recursive partitioning was also used as implemented in the R package 'rpart' (Therneau and Atkinson, 2009). Recursive partitioning models allow interactive effects between variables to be investigated.

## Major changes in land-cover as a result of land-use intensification

Important changes in land-cover/land-use types occurred during the time period analyzed within each study area, as shown by the mapping and quantification of remote sensing imagery (Fig. 2.1) and the analysis of change trajectories among land-cover types (Fig. 2.2). The observed changes between land-cover classes differed significantly among the various study areas and between the particular periods of time under study. We detected the following major trends: (1) forest degradation to shrubland (in central Chile and southern Argentina) or secondary forest (in northern Argentina and Veracruz), (2) conversion of shrubland and secondary forest to agricultural land, grassland or bare ground (in central Chile, Veracruz and Chiapas), (3) direct conversion of forest to agricultural land or grassland (in northern Argentina and Chiapas), (4) conversion during different periods between forest and shrubland and between shrubland and grassland in southern Argentina, and (5) conversion during different periods between forest and secondary forest and conversion of grassland to agriculture in Veracruz. These changes indicate overall land-use intensification across all study areas during the interval studied. Loss of natural vegetation cover, namely forest and shrubland, is the most consistently observed change. Oaxaca presented by far the most complex patterns of change trajectories, with alternate exchanges between natural vegetation cover and bare ground or agriculture over the consecutive periods of time. Transformation to urban areas was less important as compared with other changes in all areas excepting in Chiapas. Similarly, expansion of tree plantations was relatively high in central Chile and southern Argentina, but was of lesser importance in other study areas than the changes indicated above.



**Figure 2.1** Land-cover maps based on classification of remote sensing imagery from the study area in northern Argentina for the years 1976–77, 1987, 1993 and 2006, and comparison of the respective extents of land-cover classes by percentage of study area (Cristóbal *et al.*, in preparation). Quantified maps of land-cover/land-use change and of current forest cover and forest loss for all study areas can be found in Rey Benayas *et al.* (2010a and 2010b).



**Land use mosaic in La Sepultura Biosphere Reserve, Chiapas, Mexico. Photo: N. Tejedor**



**Vineyards in Casablanca valley, Chile. Photo: C. Echeverria**

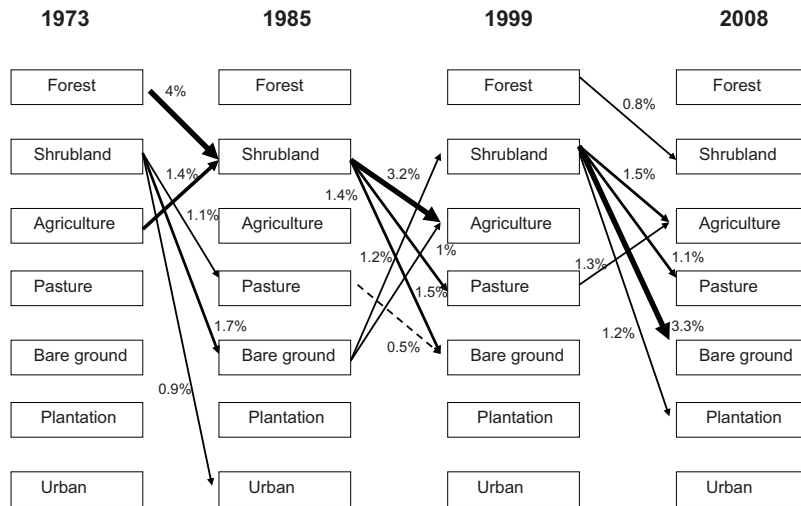


Figure 2.2a

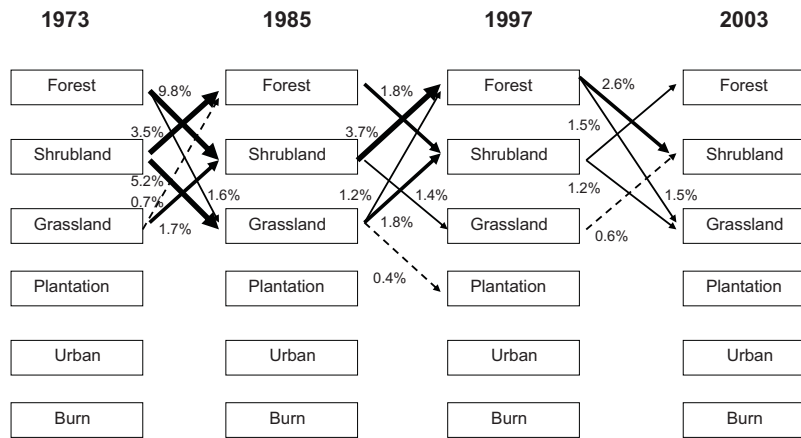


Figure 2.2b

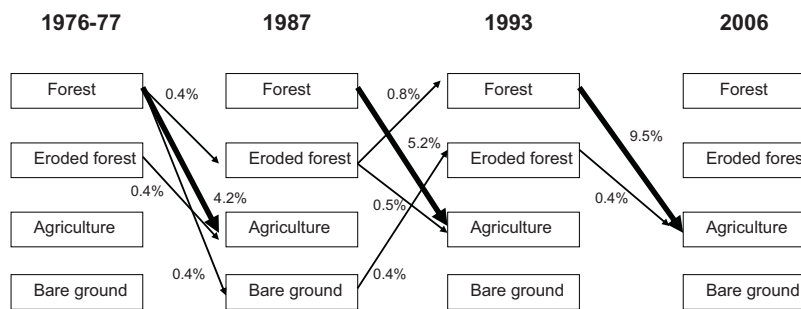


Figure 2.2c

**Figure 2.2** Major change trajectories and their contributions to net change as a percentage of the study area in (a) central Chile (thick lines correspond to a net change of  $>3.2\%$ , intermediate lines correspond to net changes between  $1.6-3.2\%$ , and thin lines correspond to net changes of  $<1.6\%$ ; only net contributions to change of  $>10,000$  ha or  $0.8\%$  of the study area are represented) (Schulz *et al.*, 2010); (b) southern Argentina (thick lines correspond to net change of  $>3.2\%$ , intermediate lines correspond to net changes between  $1.6-3.2\%$ , thin lines correspond to net change of  $0.8-1.6\%$ , and dashed lines correspond to net change of  $0.4-0.8\%$ ); (c) northern Argentina (thick lines correspond to net change  $>5\%$ , intermediate lines correspond to net changes between  $1-5\%$ , and thin lines correspond to net change  $<1\%$ ; only net contributions to change  $>3200$  hectares or  $0.4\%$  of the study area are represented); (d) Chiapas, Mexico (thick lines correspond to net change  $>0.3\%$ , and dotted lines correspond to net change  $<0.1\%$ ), ( $0.1\%$  of the study area corresponds to ca. 1600 ha); (e) Oaxaca, Mexico (thick lines correspond to net change  $>6\%$ , intermediate lines correspond to net changes between  $6-1.6\%$ , and thin lines correspond to net change  $<1.6\%$ ; only net contributions to changes  $>350$  ha. or  $0.03\%$  of the area are represented); (f) Veracruz, Mexico (thick lines correspond to net change  $\geq 3.2\%$ , and thin lines correspond to net change  $<3.2\%$ ; only net contributions to change  $>2400$  hectares or  $1.6\%$  of the study area are represented).



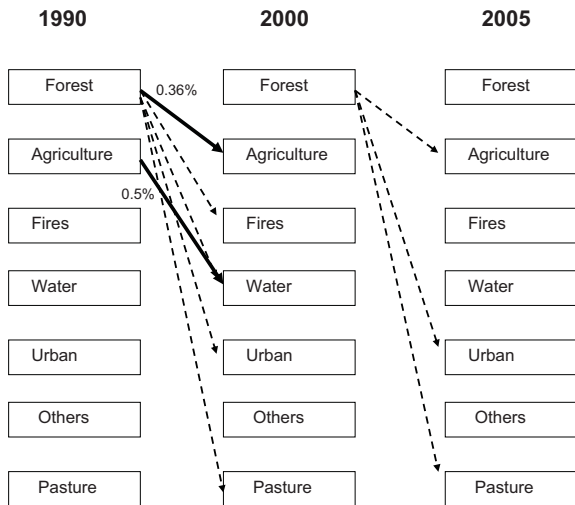


Figure 2.2d

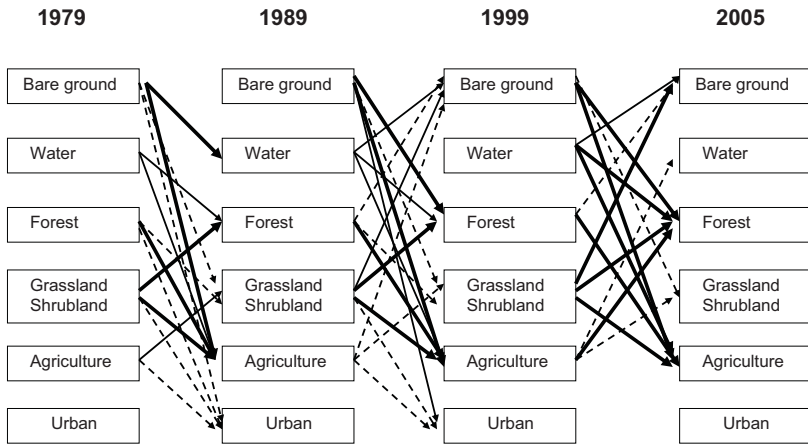


Figure 2.2e

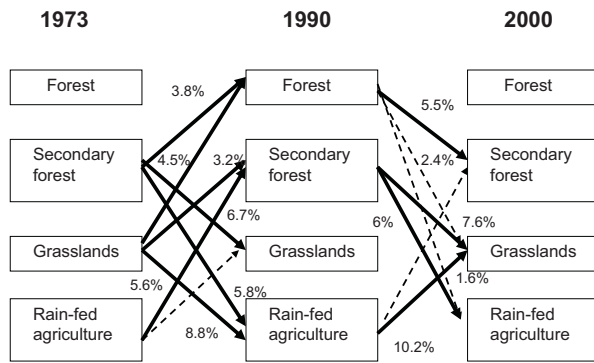


Figure 2.2f

Figure 2.2 (cont.)

Changes in land-cover/land-use types as inferred from mapping and quantification of remote sensing imagery and analysis of change trajectories among land-cover types indicate ongoing land-use intensification in all study areas. Major changes observed include a reduction in natural vegetation cover, namely forest and shrubland, and a strong increase in human-induced land-cover types such as cropland, pasture, bare ground, urban areas, and tree plantations with exotic species. Beyond this consistent pattern, the observed changes were of different intensity - when they existed at all - in the various study areas. The transformation of forest to the most highly influenced land-cover types, such as farmland, occurred with or without other intermediate land-cover types.

Land-cover change in the Mediterranean climate area of central Chile revealed a general trend towards a continuous reduction in forest and shrubland that, in turn, has led to an increase in provisioning ecosystem services such as food and timber production likely at the expense of biodiversity and hydrological services (Schulz *et al.*, 2010; **Box 2.2**). This process has involved a progressive modification from forest to shrubland vegetation, the predominant vegetation cover in this semi-arid landscape, and a relatively high loss of shrubland as a consequence of conversion to agriculture and timber plantations and, to a lesser extent, urbanization. This can be explained by an increase in local demand owing to population growth and an open market policy initiated after Chile's economic crisis at the beginning of the 1970s (Silva, 2004). The strong increase in agriculture has been stimulated by a combination of market liberalization, incentives for new export-oriented crops, introduction of new irrigation technologies, and improvements in road infrastructure (Valdés and Foster, 2005).

In contrast to central Chile, we observed an abrupt conversion of forest to agricultural land in northern Argentina, which has one of the highest agricultural land conversion rates in the country. In the 1970s, almost 95% of the area was covered by forests, and 90% by some type of tree cover. The deforested areas consisted mainly of pastures and rotational cropland plots in rural communities and small areas of premontane forests in areas where irrigation was possible. Since the 1980s, when soybean cultivation became highly profitable and began (Adelman, 1994; Brown and Malizia, 2004; Gasparri and Grau, 2009), huge areas of forest were converted at a rate of >20,000 ha/year (**Box 2.3**). Similarly in central Veracruz, Mexico, following a period of slight increase in forest area in the previous decade, forest area declined markedly in the 1990s with the establishment of powerful federal incentives, notably the Procampo programme, to promote the conversion of forest cover to cattle pasture and croplands (Klepeis and Vance, 2003; Montero-Solano, 2009). The programme originally applied to areas planted with beans, cotton, maize, rice, sorghum, soybeans, or wheat; however farmers supported by the programme could use their land for other crops, raising livestock or silviculture. Procampo is currently being phased out under the provisions of NAFTA (i.e. after 15 years) and forest cover appears to have begun to increase once again.

In Chile, there has been a pronounced expansion of timber plantations, mostly as a result of government subsidies for tree-planting that were introduced in 1974 and which stimulated the planting of *Pinus radiata* and *Eucalyptus globulus* (Aronson *et al.*, 1998). The expansion of timber plantations did not result in major conversions of native forest, as it did in southern Chile (Echeverria *et al.*, 2006) and the region in southern Argentina studied here. In this latter region, deforestation can primarily be explained by the occurrence of natural and anthropogenic fires which, in many cases, did not regenerate back into forests and remained as stable grasslands or shrublands (Mermoz *et al.*, 2005). However, dryland forest areas in this region are undergoing

## Box 2.2 Vegetation cover change in mountain ranges of central Chile (1955–2008)

C. Villablanca, J. Hernandez, C. Smith-Ramírez, J. Schulz

The sclerophyllous forest is the most characteristic plant formation in the Mediterranean climate area of central Chile. Sclerophyllous trees and shrubs have hard leaves adapted to long dry summers and occasional morning frost. This plant formation is distributed mainly on south-facing slopes and creeks, where soil humidity is concentrated. The most common tree species at the dry end of the environmental gradient of Chilean sclerophyllous vegetation are quillay (*Quillaja saponaria*), maitén (*Maytenus boaria*) and litre (*Lithrea caustica*). In more humid places such as creeks and sheltered slopes, trees such as peumo (*Cryptocarya alba*), patagua (*Crinodendron patagua*), belloto del norte (*Beilschmiedia miersii*), pitra (*Myrceugenia exsucca*) and canelo (*Drimys winteri*) are found (Donoso, 1995).

Chilean sclerophyllous forests have high species richness and endemism (Villagrán, 1995). The high endemism and increasing degree of threat owing to extensive land-cover change have resulted in the inclusion of these forests as one of the 25 global biodiversity hotspots (Myers *et al.*, 2000). However, economic activities and increasing human population in central Chile have resulted in high impacts on natural resources, leading to important losses of biodiversity. Sclerophyllous forests have been eradicated and degraded over large areas of central Chile, especially in the central valley (see **Box 2.4**), but also in the Andean and coastal ranges. Deforestation and land use change are the result of expanding farmland and silvicultural plantations, as well as urban expansion (Armesto *et al.*, 2010; Schulz *et al.*, 2010). At the same time remnant stands are used for firewood and soil extraction, cattle grazing and trampling, and most importantly, subject to a high frequency of anthropogenic fires. Herbivory of shrub and tree seedlings by rabbits, horses and goats, is a major factor preventing sclerophyllous forest recovery on abandoned lands (Fuentes *et al.*, 1983).

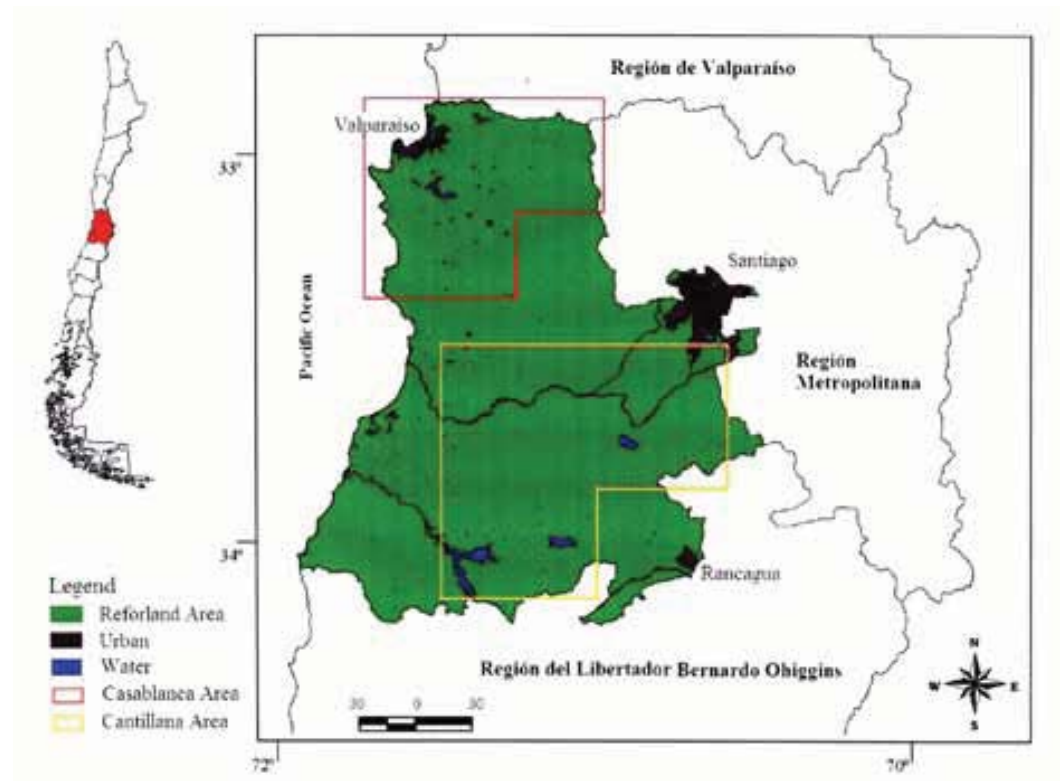
The objective of this study was to assess the changes in land-cover that have occurred in the second half of the twentieth century in the Mediterranean-climate region of central Chile. We chose the period 1955–1975, as an antecedent to the patterns described by Schulz *et al.* (2010) for the period 1975–2008 in the same region. The study area was the landscape surrounding the Casablanca hills and valleys (2740.2 km<sup>2</sup>, 32°50'00"–33°27'00" S and 71°36'00"–70°58'00" W (from sea level to 2190 m a.s.l.) and Cantillana hills (4304.18 km<sup>2</sup>, 33°38'00"–34°15'00" S and 71°27'00"–70°38'00" W, from 145 to 2280 m a.s.l.), (**Fig. 1**). Plant species richness and composition were documented in sclerophyllous vegetation patches larger than 60 ha. We excluded extensive areas that had been deforested prior to 1955. Such areas are found south and east in the valley between Casablanca and the Cantillana hills. We estimated vegetation cover in 1955 using aerial photographs, and distinguished eight different cover classes: forests, shrubland, farmland, urban areas, bare soil, anthropogenic prairies, and exotic silvicultural plantations. Subsequently, we compared our results for 1955 with the vegetation cover map for 1975, obtained from Landsat images. We identified and quantified the major changes by cover type during this 20-year period.

To estimate land-cover changes between 1955 and 1975 we applied the Land Change Modeller in the IDRISI software programme; we used Landsat images to estimate land-cover changes between 1975 and 2008. The changes reported here are notable because shrublands in central Chile have been considered extremely persistent. The dynamics of land-cover change over these two decades of the twentieth century is shown in **Table 1**, **Figs. 2** and **3**. Between 1955 and 2008, 71,290.75 ha of forest were lost. From the total forest cover that existed in 1955, 64.2% (14,3160.96 ha) was below 700 m. Plant species richness is higher in Cantillana, and in Casablanca below 600 m (Universidad de Chile, 2009). However, the areas of greatest species richness were replaced by other land-cover types by the first half of the twentieth century or earlier.

The percentage of change in sclerophyllous forest cover from 1955 to 1975 was comparatively low compared to the period 1975 to 2008, where natural vegetation cover was lost much more rapidly (Schulz *et al.*, 2010). In the study areas, between 1955 and 1975, forest cover decreased by 8.5%, however between 1975 and 2008, forest cover decreased by 45%. Between 1955

**Box 2.2 (cont.)**

and 1975 Matorral (shrubland cover) increased by 5.5%, however, it decreased by 22.7% in the period between 1975 and 2008. This was probably the result of the expansion in agricultural land and the development of an agricultural export industry in Chile coinciding with the opening up of international markets. The development of new methods of irrigation during the 1980s was responsible for the expansion of cultivated land up to altitudes of 500 m on coastal hills; this affected the plant diversity of many remnant patches of sclerophyllous forest and shrubland.

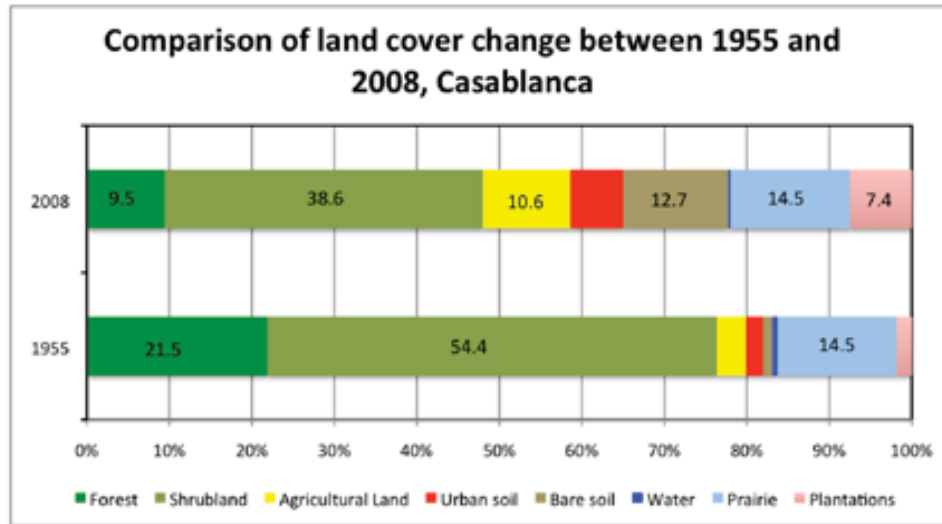


**Figure 1** Area of study in the Casablanca and Cantillana hills (red and yellow lines indicate the area mapped).

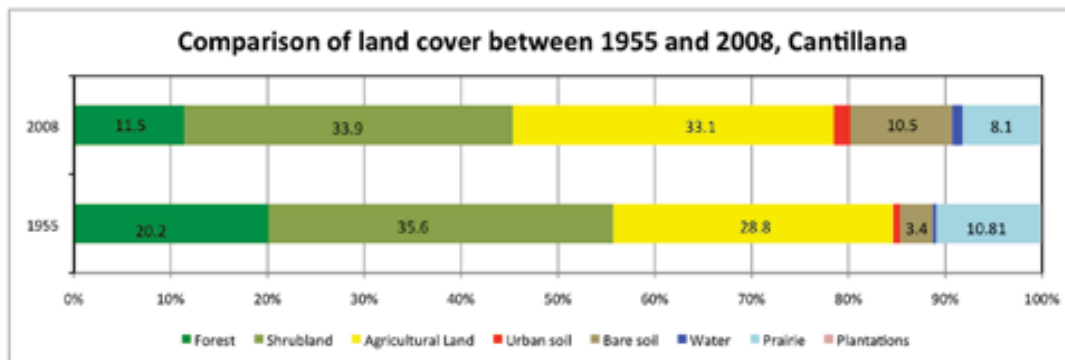
**Table 1** Dynamics of land-cover change in hectares on the Casablanca (CB) and Cantillana (C) sites.

Ha	Forest	Shrubland	Agricultural land	Urban soil	Bare soil	Water	Prairie	Plantations
CB 1955	59,905.51	148,625.93	9,634.07	5574.85	3120.80	1690.97	39,440.08	5031.51
CB 2008	25,549.62	104,176.17	28,550.01	17,471.1	34,329.55	801.46	39,043.49	20,238.92
C 1955	83,255.45	147,079.46	119,254.14	3062.85	14,056.02	1425.32	44,688.32	401.11
C 2008	46,320.59	136,973.54	133,873.81	6971.57	42,607.54	4115.66	32,685.24	578.71

**Box 2.2 (cont.)**



**Figure 2** Land-cover change in Casablanca between 1955 and 2008.



**Figure 3** Land-cover change in Cantillana between 1955 and 2008.

**Box 2.3** Land-use change in the Yungas Biosphere Reserve and its area of influence, Argentina (1975–2008)

*S. Pacheco, L. Cristóbal, K. Buzza*

The Biosphere Reserve of the Yungas (RBYungas) was created in northwestern Argentina in 2002 under the UNESCO Man and the Biosphere (MAB) programme. It is one of the largest reserves in the country, extending over approximately 1,350,000 ha, and includes two provincial territories (Jujuy and Salta). The reserve was created as part of efforts to implement actions conducive to conserving and managing the Yungas region (subtropical mountain forests) in a sustainable way.

The RBYungas includes mainly subtropical mountain forests, particularly in the northern latitudinal sector, which are functionally connected to the central sector of the Yungas and to the Chaco forests in the surrounding areas. As a result of agricultural activity, these environments are being fragmented, with remaining fragments sometimes connected by areas that act as corridors. The objective of this study was to describe the land-use change process in the northern and central sectors of the Yungas and its transition to Chaco, with special emphasis on the RBYungas for the period 1975–2008.

### Box 2.3 (cont.)

The identification of transformed areas was carried out through visual interpretation of Landsat satellite images. We developed a time series of an area of more than 5 million ha located in Jujuy and Salta. The years included were 1975, 1985, 2005 and 2008. For the four years analyzed, we calculated the total deforested area and the annual transformation rate. For each year of the time series, we identified types of crops, determined their slope and established the ecoregion that was transformed in each case. For the RBYungas we determined the deforested area for each year analyzed and the remaining surface which may be subject to transformation.

From 1975 to 2008, the transformed surface in the study area increased almost 13% (**Table 1**). During the 1970s, the transformed areas were mainly concentrated in the flat areas of the premontane forest, on the west side of the study area. During the 1980s, the expansion of the agricultural frontier began in the east of the region, mainly in the province of Salta, occupying the Chaco environments (**Fig. 1**).



**Figure 1** Spatial distribution of transformed land in the study area and RBYungas for the years 1975, 1985, 2005 and 2008.

The productive activities carried out in the 1970s include sugar cane, tobacco and agricultural cultivation. During 1985 these three activities remained important but the production of soybean combined with beans represented almost 30% of production. During 2005, these two new categories accounted for almost 50% of production. During the four years analyzed, 90% of the transformation occurred on slopes below a 5% gradient. The transformation of the forest in steeper areas was mainly for forest plantations, pastures and agricultural plots of small size.

### Box 2.3 (cont.)

**Table 1** Transformed area in different years and annual transformation rate for the study area during the period 1975–2008.

Year	Transformed (ha)	% of RBYungas	Annual transformation rate (ha)
1975	359,143	6,7	
1985	611,295	11,5	25,215 (1975–1986)
2005	908,845	17	14,878 (1998–2004)
2008	1,034,486	19,4	41,880 (2004–2008)

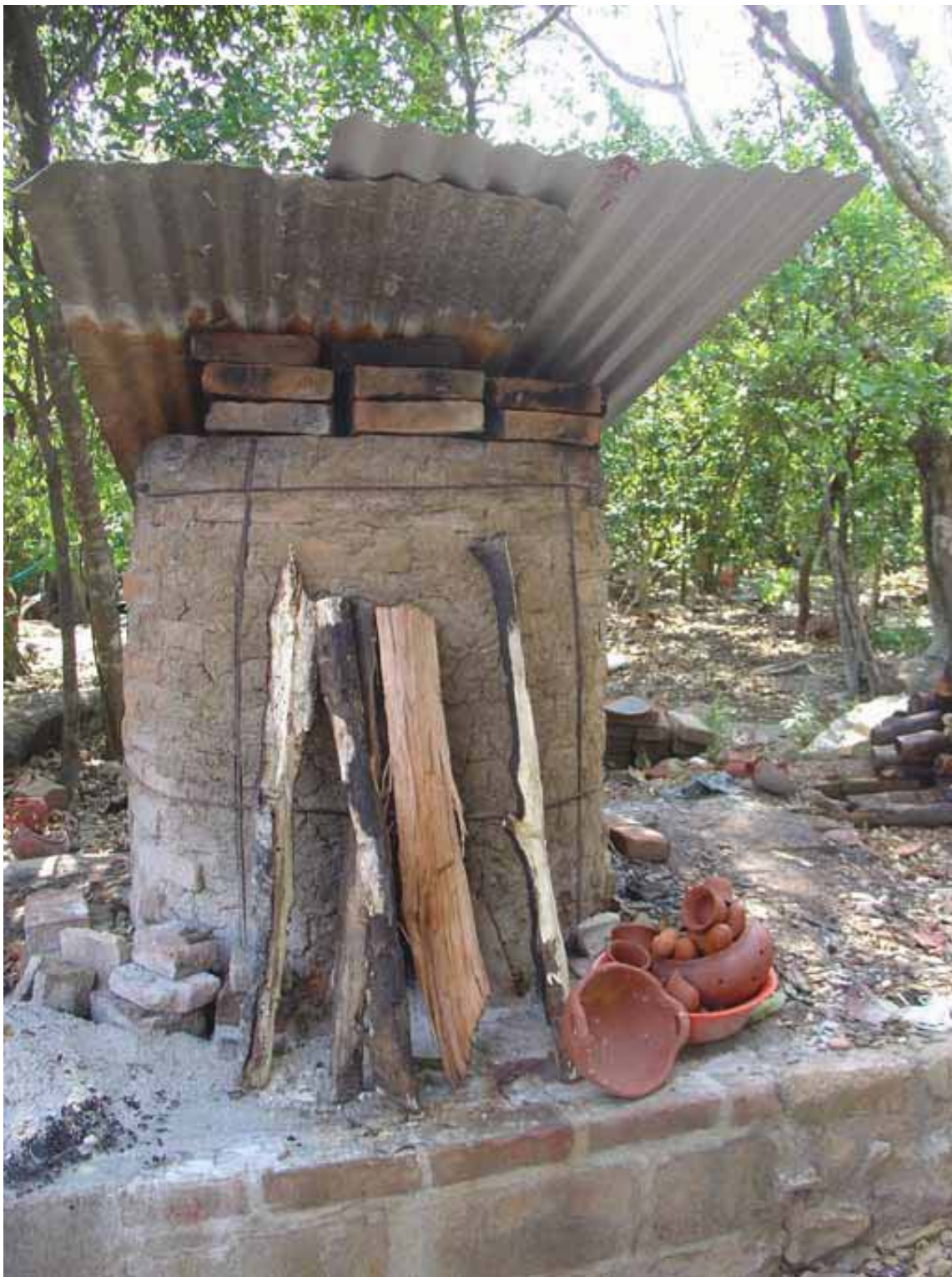
### Transformation within RBYungas

In 1975, 4% of the RBYungas was transformed into agricultural land, mainly concentrated in flat areas; this reached 6.5% in 2008. This represents an annual transformation rate which increased from 930 ha for the period 1975–1985 up to 3274 ha between 2005 and 2008 (Table 2). Almost 90% of the RBYungas surface is represented by steep areas with a slope of a gradient of more than 5%. Only 150,000 ha correspond to flat areas suitable for agricultural use. If we analyze the transformed area taking into account only available flat areas within the RBYungas, we observe that during 2008 more than 50% of this surface was transformed (Table 2).

**Table 2** Transformed surface in different years and annual transformation rate for the RBYungas during the period 1975–2008.

Year	Transformed (ha)	% of RBYungas	% flat areas in RBYungas	Annual transformation rate (ha)
1975	53,597	4.0	34.2	
1985	62,725	4.7	39.4	930 (1975–1986)
2005	77,164	5.7	49.2	722 (1998–2004)
2008	86,987	6.5	55.3	3.274 (2004–2008)

These results indicate that the transformation of premontane forest has been increasing in recent years. This process began in areas with slopes of less than 5% gradient with the planting of sugar cane which requires irrigation; it resulted in the near disappearance of forests in flat areas at a regional level. Since the 1980s, there has been an expansion of the agricultural frontier towards drier areas of the Chaco owing to changes in technology, an increase in precipitation in the last century and the incorporation of the soybean crop which does not require irrigation. More than 50% of the flat surface of the RBYungas has already been transformed into agricultural land. RBYungas remains connected to its surrounding natural areas through those slopes that remain forested. Currently, local governments have implemented land-use plans, which delimit production areas and areas that must be protected in order to maintain the functional connection between the different environments of the region.



**Firewood next to a potter's kiln, Chiapas, Mexico. Photo: B. Ferguson**



a small but steady conversion to plantations with exotic conifers. In southern Argentina, rapid expansion of urban areas coincided with the abolition of the urban limits by the Ministry of Housing and Urbanism in 1979 and the liberalization of the urban land market (Kusnetzoff, 1987).

## Changes in forest extent

Forest loss was consistently detected in all study areas, ranging from an annual rate of -1.7% in central Chile to a negligible -0.12% in the Central Valley of Chiapas, with an average rate of -0.78% across all study areas (Table 2.1). Some study areas have experienced a relatively high proportion of forest loss (15% to 9% of the study area in central Chile, 90% to 70% in northern Argentina, and 11.3% to 6.56% in central Veracruz), whereas others have experienced relatively little deforestation (13% to 11% in southern Argentina, 59% to 57% in Oaxaca, Mexico) or hardly any changes in their forest extent (32% of the study area in Chiapas at both reference dates, as 68% of forest land had already been lost in this region by the beginning of the study period).

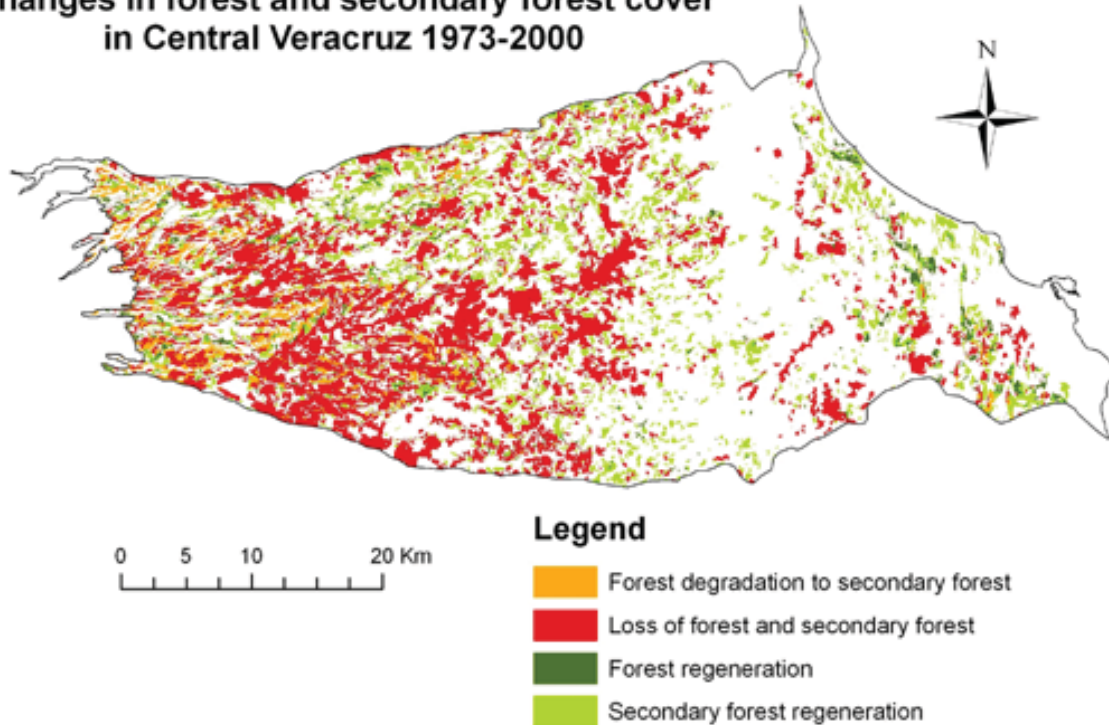
**Table 2.1** Percentage forest cover detected in the 1970s (1990 for Chiapas\*) and mid-2000s (see Table 1 Box 2.1 for exact dates) and annual deforestation rates in each study area. \*\*Refers to undisturbed forest cover in the 1973–2000 period when images are all Landsat and thus have more comparable spatial resolutions.

Study area	% Forest cover in early or mid-1970s	% Forest cover in 2000s	% Annual rate of change
Central Chile	43.3	33.9	-1.7
Southern Argentina	17.3	16.4	-0.17
Northern Argentina	94.0	73.0	-1.3
Veracruz (Mx)	11.3	6.56	-1.22**
Oaxaca (Mx)	59.3	56.6	-0.18
Chiapas (Mx)	32.1*	31.5	-0.12

However, forest loss varied considerably between the analyzed time periods in most study areas. Thus, northern Argentina showed high forest loss (>5%) between all analyzed periods, central Chile lost forest extent in all periods except from 1985 to 1999 when there was hardly any change, and central Veracruz gained total forest cover at a yearly rate of 0.6% between 1973 and 1990 but then underwent considerable forest loss at a yearly rate of 4.33% between 1990 and 2000.

The analysis of change trajectories revealed that the patterns of conversion of forest cover also differed between study areas. The conversion of forest to agriculture was mediated by an intermediate shrubland state in central Chile (Fig. 2.2) and southern Argentina, whereas in Veracruz, a similar trend was observed for the conversion from primary forest to agriculture mediated by an intermediate secondary forest state. However, in northern Argentina and Oaxaca, and to a lesser extent in Chiapas, most forest loss occurred as a result of direct transformation into agricultural land. In all study areas, loss of forest extent was partially mitigated by forest re-growth as a result of farmland abandonment (Fig. 2.3).

### Changes in forest and secondary forest cover in Central Veracruz 1973-2000



**Figure 2.3** The extent and distribution of primary forest at a given time is a balance between forest loss and degradation and forest regeneration. This figure illustrates the distribution of these different processes for the case study landscape in Veracruz, Mexico, for the period 1973–2000. Similar maps for all study areas can be found in Rey Benayas *et al.* (2010b).

While forest loss was a consistently observed outcome of land-use intensification, annual rates of deforestation were highly variable between study areas and among different time periods within the same study area. The highest deforestation rate was observed in Chile; nevertheless, this rate is relatively low compared to rates in temperate forests in south-central Chile (Echeverria *et al.*, 2006). This is probably attributable to the fact that central Chile has been densely populated since early European colonization and major conversions of forest cover had taken place long before the 1970s (Conacher and Sala, 1998; **Box 2.4**). Historical data from central Veracruz suggest that dryland forest cover has been suppressed (<33% cover) at least since the start of the previous century when land reforms were implemented following the Mexican revolution (**Box 2.5**). Similarly, most deforestation in Chiapas occurred long ago (Challenger, 1998) and has been relatively minor in recent times (**Box 2.6**). Our results suggest that this region has undergone a comparable level of historical deforestation to other dry forest ecoregions in Mexico. Challenger and Dirzo (2009) reported that dry forest loss at the country level between 1976 and 1993 amounted to 177,000 ha per annum (annual deforestation rate of 1.6%) and reduced to 44,416 ha per annum (annual deforestation rate of 0.5%) over the next decade (1993–2002). Deforestation rates found in our study area in Oaxaca were slightly lower as compared with other areas with similar vegetation in Oaxaca and in Mexico (Aguilar *et al.*, 2000; Velázquez *et al.*, 2003; Díaz-Gallegos *et al.*, 2008). Cayuela *et al.* (2006), however, reported an annual deforestation rate of 4.8% for the highlands of Chiapas in the period 1990–2000.

Primary forests are lost because they are directly converted to cropland or grazing land or because they are degraded by permanent grazing pressure, firewood collection and charcoal

## Box 2.4 Historical distribution of the dryland forest in central Chile during the Spanish conquest in the 16<sup>th</sup> century

C. Echeverría, R. Fuentes, R. Torres, P. Camus.

Historic landscape reconstruction through documentary sources is useful to (i) assess the dynamics of historical land-use changes, (ii) define the potential of dryland forest ecosystems, and (iii) place resource management practices of indigenous people and other local communities in a historical context (Prieto *et al.*, 2003). The coastal zone in central Chile that extends from the cities of Santiago to Valparaíso presents evident signs of environmental degradation that has occurred since the 16<sup>th</sup> century. Traditionally, Chilean historians have presented an idyllic image of a territory covered by beautiful forests (Barros Arana, 1884), while others, more recently, have provided information indicating an absence of forests over large areas in historic times (Camus, 2002). Primary documentary sources offer a valuable description of the past as they narrate the first impressions of European colonists when they laid eyes on the landscape for the first time.

The reconstruction and mapping of dryland forest in the 16<sup>th</sup> century, at the onset of European colonization, is key to a clear understanding of the current and historical patterns of land-use change and the development of restoration strategies in central Chile. The objective of this exercise was to reconstruct a picture of the vegetation between Santiago and Valparaíso through a spatially-explicit approach that integrates information from documentary sources and environmental factors into a GIS.

Visual descriptions of vegetation were obtained from field notes taken from travellers through the region. These descriptions were collected from reviews of primary and secondary historical sources written in the 16<sup>th</sup> century. Most of the visual descriptions used in the present study were gathered in the Casablanca valley, Colina zone and the Aconcagua River route. The main primary documentary sources used in this study were drawn from registries maintained by Santiago's Cabildo (town council), land measurements and registries, chronicles, letters and travel diaries. However, further descriptions from secondary documentary sources were also used when they clearly referred to descriptions of vegetation that existed before the arrival of the Spanish. Visual descriptions of naturalists such as Charles Darwin and Edward Poeppig were used to model species' distributions on the Aconcagua-Valparaíso route and its surroundings. We discarded those descriptions that did not have a precise enough spatial reference. As a result, we did not use Claudio Gay's botanical descriptions.

The descriptions and roads used by travellers were spatially plotted on a 30 m-resolution elevation map. Then, environmental requirements such as aspect and elevation of the description points in relation to vegetation composition were obtained through a review of the literature (Donoso, 1982; Donoso, 1995). This information was used to generate digital maps of habitat suitability for each category of vegetation (group of species or individual species in some cases) through a combination of ranges of elevation with categories of aspect (Table 1). North and south aspects are the major environmental factors that determine patterns of species distribution in dryland landscapes in this part of Chile (Donoso, 1995). South-facing hills are characterized by lower levels of solar radiation/insolation, and therefore higher humidity in the soil and air than north-facing sites.

In some cases the documentary sources provided specific locations for some currently threatened species such as *Jubaea chilensis* (Grau, 2004) and *Porlieria chilensis*. This enabled the historical distribution at the species level to be mapped. Similarly, several historical descriptions mentioned the abundant presence of *espinales*, a disturbed pseudo-savannah dominated by *Acacia caven*, across the study area. This enabled different sub-categories of *espinales* to be mapped. Additionally, documentary sources containing detailed descriptions of some of the current main cities in the study area such as Santiago, Valparaíso and Quillota were mapped.

## Box 2.4 (cont.)

**Table 1** Equivalence of meanings and historical descriptions for each detected vegetation category.

Vegetation type	Altitude (m a.s.l) and aspect	Sample of original texts described by travelers during the 16 <sup>th</sup> century	Historical references
<i>Grassland</i>	> 200 North and East	"Las únicas plantas que cabría mencionar eran arbustos pequeños e inaperentes, que ostentaban sólo de vez en cuando todavía en temporada desfavorable una miserable flor y que se presentaban semisecos y polvorientos"	Poeppig, E. 1835
<i>Open espinal</i>	200 - 350 North and East	"... Todos los campos estaban desiertos, solo se veían cubiertos de ciertos arboles espinosos que hacen muy incómodo el camino".	Frezier, M. 1716
	> 300 South and West		
	350 - 450 North and East	"... me encuentro otra vez en tierras salvajes cubiertas en forma muy rala por acacias y algarrobos cuya compañía, empezaba yo a sospechar, no iba a perder en esta tierra"	Schidtmeyer, 1824
<i>Dense espinal</i>	300 - 350 South and West		
	350 - 450 North and East	"... Numerosos troncos mejor talados a uno o dos pies del suelo, donde ensanchaban su espacio estéril no parecían mejorar su perspectiva, pero indicaban que este valle había estado cubierto tupidamente por ellos antiguamente ... "	Schidtmeyer, 1824
	300 - 350 South and West		
	350 - 450 North and East	"... noto sobre la vertiente septentrional no crecen sino zarzas ... "	Darwin, C. 1839
<i>Sclerophyllous forest</i>	300 - 350 South and West		
	350 - 450 North and East	"... He visto algunos lugares bonitos, que consisten en pequeñas colinas y cañadas de formas suaves, cubiertas de varias clases, mas vegetación que la vista hasta ahora y de verdor ás agradable"	Schidtmeyer, 1824
	350 - 1000 South and West		
	350 - 450 North and East	Santiago "... es un hermoso y grande llano como tengo dicho. Tiene a cinco y seis leguas montes de muy buena madera que son unos árboles muy grandes que sacan muy buenas vigas. hay otros árboles que se llaman canela"	Gerónimo de Bibar, 1558
	350 - 1000 South and West		
	350 - 1000 South and West	Estero de Pocochay como "Maquilemu", o "bosque de maquis".	Ginés de Lillo, 1605
	350 - 1000 South and West	"... la vertiente meridional está cubierta de un bambú que llega a alcanzar hasta 15 pies de altura".	Darwin, C. 1839

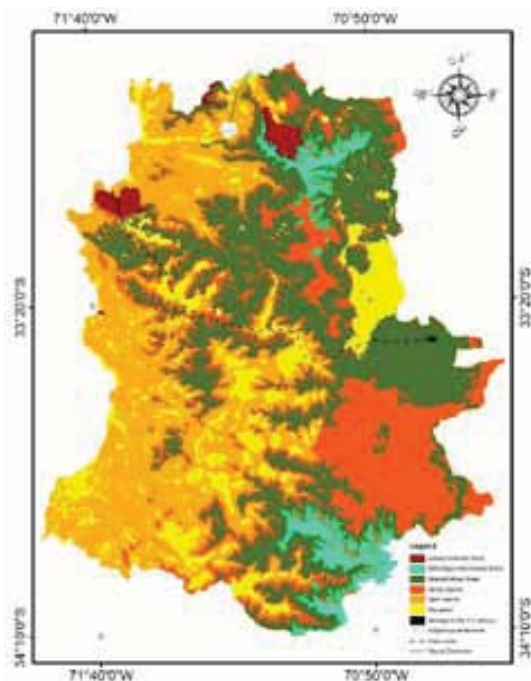
**Box 2.4 (cont.)**

**Table 1** (cont.)

Vegetation type	Altitude (m a.s.l) and aspect	Sample of original texts described by travelers during the 16 <sup>th</sup> century	Historical references
<i>Jubaea chilensis</i>	500 - 2000 North	"Hay palmas y solamente las hay en esta gobernación en dos partes, que es en el río de Maule hay un pedazo que hay de estas palmas, y en Quillota las hay en torno de siete y ocho leguas"	Gerónimo de Bibar, 1558
	500 - 2000 North	"En algunos lugares se encuentran palmeras y quedo muy asombrado al hallar una de ellas a 4.500 pies de altitud ... "	Darwin, C. 1839
<i>Indigenous settlements</i>	>200	"La población vivía sobre todo en ese Sector, llamado también 'Camino de Coquimbo', pues este seguía por Calera a la cuesta del Melón y se juntaba en La Ligua con el de la costa, para continuar al Norte".	Keller, Carlos, 1960

The reconstruction (mapping) of the vegetation revealed that there was once a greater number of species spread over a greater area (**Fig. 1**). In the 16<sup>th</sup> century, sclerophyllous forest occupied an area of approximately 115,000 ha, mainly south facing. *Espinales* covered approximately 670,000 ha on north-facing and flat sites. At the municipal level, 21% of the total area in Casablanca municipality was occupied by sclerohyllous forest and 66% by *espinales*. In Quilpué, these values were 18% and 68%, respectively; and in Melipilla, 34% and 67%, respectively.

At the species level, the reconstruction of *J. chilensis* distribution revealed that this once covered approximately 17,000 ha, a much larger area than at present (**Fig. 1**). The species was distributed in two main populations, one on the coast around the current city of Viña del Mar, and the other in the La Campana mountains (**Fig. 1**). *Nothofagus macrocarpa* forest covered most of the summit on the Roble hills and the Cantillana hills (**Fig. 1**), with a total area of 33,800 ha. Towards the east, *Porlieria chilensis* occupied an area of approximately 5,100 ha of the hills around the Santiago area (**Fig. 1**).



**Figure 1** Distribution of vegetation types and human settlements during the Conquest in the 16<sup>th</sup> century.

### Box 2.4 (cont.)

Although the area of forest coverage in 16<sup>th</sup> century was larger as compared to the present day, in the eyes of the newly arrived Spanish population of central Chile, tree species were already scarce at that time. The Cabildo therefore issued several directives regulating the cutting of trees. One of the main indigenous settlements in the 16<sup>th</sup> century was established in the current Quillota valley and occupied approximately 1,800 ha (**Fig. 1**). Santiago was the main Spanish settlement during the Conquest, extending over 320 ha (**Fig. 1**). Some documentary sources also provide evidence of the main disturbances affecting vegetation in central Chile in the 18<sup>th</sup> century after the founding of Santiago. The establishment of Santiago resulted in the expansion of urban areas and rangelands for livestock into the Central Valley during the 18<sup>th</sup> century. This expansion resulted in the high consumption of tree and shrub species (Cunill, 1995).

Our results demonstrate that by the 16<sup>th</sup> century, the landscape in central Chile was dominated by different vegetation types. Sclerophyllous forest and some species were more abundant on south-facing slopes, while *espinales* covered large areas on the north-facing and flat sites across the study area. The presence of this human-induced vegetation type reveals that the original vegetation had already been disturbed by indigenous people. All this indicates that dryland vegetation has been profoundly and irreversibly transformed since the Conquest until the present time, highlighting the need for ecological restoration. The historical analyses presented here can be used to inform restoration plans.

### Box 2.5 Historical reconstruction of land-use patterns from 1920 to 1960 on communal lands of Paso de Ovejas, Veracruz, Mexico

J. Ortiz, F. López-Barrera, J. Callejas, R.H. Manson

The primary forests in the municipality of Paso de Ovejas suffered few alterations following the Spanish conquest. However, in the 19<sup>th</sup> century these lands came under the management of the military, merchants and foreign investors. Most forest cover in the lower sections of the municipality, where irrigation was increasingly prevalent, were subsequently replaced by commercial plantations such as sugarcane. Conversely, the highlands, hills, slopes and canyons, which were largely isolated from human infrastructure and roads, suffered relatively few alterations to their natural vegetation cover until the first decades of the 20<sup>th</sup> century, when they were used as pastures.

Increases in human populations were also a contributing factor to these changes. In 1799, census data registered a population of barely 100 people. A century later this number had increased to 3572. The heads of family included men who worked on the land of the large private properties (*haciendas*) as labourers, sharecroppers, and day labourers. Each family had permission to use a small part of the cultivated land to grow staples such as corn, beans and chillies, and to raise domestic animals such as pigs or chickens.

After the Mexican revolution and the establishment of the Constitution of 1917, these *haciendas* were largely dismantled as part of agrarian reforms. Initially, the most productive lands were transformed into communal lands known as *ejidos* and divided among the peasants that had worked in the *haciendas* for many years. Under the *ejidos* systems land was kept in collective trusts for the peasant communities who were allowed to use it for farming and natural resource extraction. In the first thirty years of the 20<sup>th</sup> century, the population of Paso de Ovejas doubled to more than 7350 people. This population growth continued during the next two decades. By the 1970s, owing to both intrinsic population growth on communal lands and the arrival of new immigrants, the population had doubled again reaching a total of 15,271 people. The greatest increases were observed among the male population, specifically the 25 to 29-year age group.

## Box 2.5 (cont.)

The combination of rapid population growth on communal lands, including those used for cultivation and cattle ranching, and those considered too infertile for agriculture, put considerable strain on this new form of land tenure. The *ejido* system is one of the main legacies of the Mexican revolution; it was incorporated in the 1917 Constitution. *Ejido* members lived in communities. They were designated land for housing with separate parcels of land designated for cultivation. As family sizes increased, the parcels of land they were assigned came under increasing pressure resulting in intensification and transformation of land uses. Forests came under particular pressure as they were felled for fuel or lumber, and in order to make room for crops, and cattle pastures.

In addition to these changes in land ownership and population demographics, the conversion of forest cover to other land uses was actively promoted by public policies, and related financial and technical assistance provided to *ejidos* by the State and Federal governments. In 1920 the Law for Idle Lands was established nationally and triggered large-scale deforestation across Mexico. The objective behind this law was to increase the volume of crops to feed the rapidly growing Mexican population. This was followed by the creation of the National Ejidal Credit Bank (1935), the organization of farmers in the National Peasant Confederation (1938), the creation of the Agriculture Bureau and Mexican Fertilizers Bureau (1943), the establishment of the National Agriculture Plan (1953), and the formation of the National Seed Producers Organization (1960), all of which facilitated the conversion of forest cover to other land uses.

As a result, remaining forest cover was increasingly limited to inaccessible lands (steep slopes or rocky soils far from roads and towns). Land-use patterns for 23 *ejidos* in the municipality of Paso de Ovejas from 1927 to 1968 are described in Table 1. These data were obtained through the revision of historical documents and maps from the National Agrarian Registry in Mexico (Registro Agrario Nacional). Forest cover was found to be absent (13 *ejidos*) or limited (from 19% to 35%) in most *ejidos* with only two showing forest coverage of more than 40% of their land area (mainly secondary forests; Table 1).

This study showed that in some tropical areas in Mexico, most deforestation, degradation, and fragmentation of forest cover probably occurred prior to 1920. These patterns of land-use change are largely undetectable using current satellite-based methods and imagery data available since the early 1970s. For example, in Paso de Ovejas, the extension of primary forest was only 2.06% in 1973 and increased to 6.28% in 1990 (Montero *et al.*, in prep). These results highlight the importance of a long-term historical perspective for understanding and interpreting current patterns of land use and the overall impact of public policies on patterns of land-use change in the region.

**Table 1** Historical records of type of land use present on 23 communal lands (*ejidos*) in central Veracruz, Mexico, including the year when they were created, area, and percentage cover of different types of land use.

<i>Ejido</i> name	Year	Extension (ha)	Rain-fed agriculture	Irrigated agriculture	Grasslands	Primary and secondary forest	Urban
Acazónica	1927	1610	14.7		72.9	12.4	
Tierra Colorada	1928	150	100				
Plan de manantial	1928	312	79.2		20.8		
Palmaritos	1928	482	24.1	52.5	23.4		

## Box 2.5 (cont.)

**Table 1** (cont.)

<i>Ejido</i> name	Year	Extension (ha)	Rain-fed agriculture	Irrigated agriculture	Grasslands	Primary and secondary forest	Urban
Paso de Ovejas	1930	1617	100				
Bandera de Juárez	1930	472	11.0		10.6	78.4	
Loma del Nanche	1930	262	64.9			35.1	
Puente Julia	1930	168	100				
Cerro de Guzmán	1931	359	54.3		21.4	24.2	
Mata Grande	1931	100	76.0		24.0		
Paso Panal	1932	370	100				
Patancán	1932	180	100				
La Víbora	1935	279	100				
Cantarranas	1935	1124	66.2			33.8	
El Angostillo	1936	900	80.0			20.0	
Cocuyo	1936	428	80.4			19.6	
El Mango	1937	211	100				
Mata Mateo	1937	714	52.9			47.1	
Mata Grande	1941	260	67.7		32.3		
Rancho Nuevo	1958	630	82.5		15.9		1.6
El Angostillo	1964	900	75.2			22.6	2.2
Acazónica	1964	2182	48.1		40.6	9.2	2.1
Loma del Nanche	1968	156			100		
<b>Average</b>		<b>602.87</b>	<b>68.57</b>	<b>2.28</b>	<b>15.74</b>	<b>13.15</b>	<b>0.26</b>
<b>Standard deviation</b>		<b>554.98</b>	<b>31.17</b>	<b>10.94</b>	<b>25.85</b>	<b>20.09</b>	<b>0.69</b>





*Austrocedrus chilensis* stand, Nahuel Huapi, Argentina. Photo: A.C. Newton



Shrubland and steppe vegetation, Nahuel Huapi, Argentina. Photo: J. Birch

## **Box 2.6** Tuning up coarse-grained potential vegetation maps for estimation of historical forest loss in tropical Mexico

*R. Vaca, L. Cayuela, J.D. Golicher*

In areas with a long history of disturbance, historical forest loss is a major issue. Most deforestation in these areas has occurred prior to the development of remotely sensed techniques. In such circumstances potential vegetation maps can be used as a baseline for the estimation of historical forest loss (e.g. Trejo and Dirzo, 2000), as they represent the area hypothetically covered by forest in the absence of human disturbance (Bredenkamp *et al.*, 1998; Moravec, 1998). However there is a general concern that the resolution of most of the available maps of potential vegetation is too coarse for real-world applications (Bredenkamp, *et al.*, 1998; Hartley *et al.*, 2004).

A typical method for constructing a potential vegetation map involves identifying remnants of vegetation with natural or near-natural character (Zerbe, 1998). The vegetation found in these remnants may be assumed to potentially extend to a wider geographical area with similar environmental conditions (Moravec, 1998; Zerbe, 1998). Map inaccuracies often result from the coarse resolution in the available maps of predictor variables (van Etten, 1998). Coarse scale maps may overlook variability in mountainous and other areas in which fine scaled climatic gradients determine the observed vegetation type (Franklin, 1995). In order to use vegetation maps effectively their resolution must be adjusted to the needs of pure and applied biological and ecological research (Araújo *et al.*, 2005; McPherson *et al.*, 2006). For many applications, data at a fine ( $\leq 1$  km<sup>2</sup>) spatial resolution are necessary to capture environmental variability that can be partly lost at coarser-grained resolutions (Hijmans *et al.*, 2005). A particular challenge is thus the generation of a fine-grained map of the potential vegetation over a large area. Climate is widely known to condition the formation of different vegetation types (Woodward, 1987). Thus, when the grain of potential vegetation maps is coarser than that of climatic layers, one possible solution is to use climate to downscale potential vegetation maps through statistical modelling.

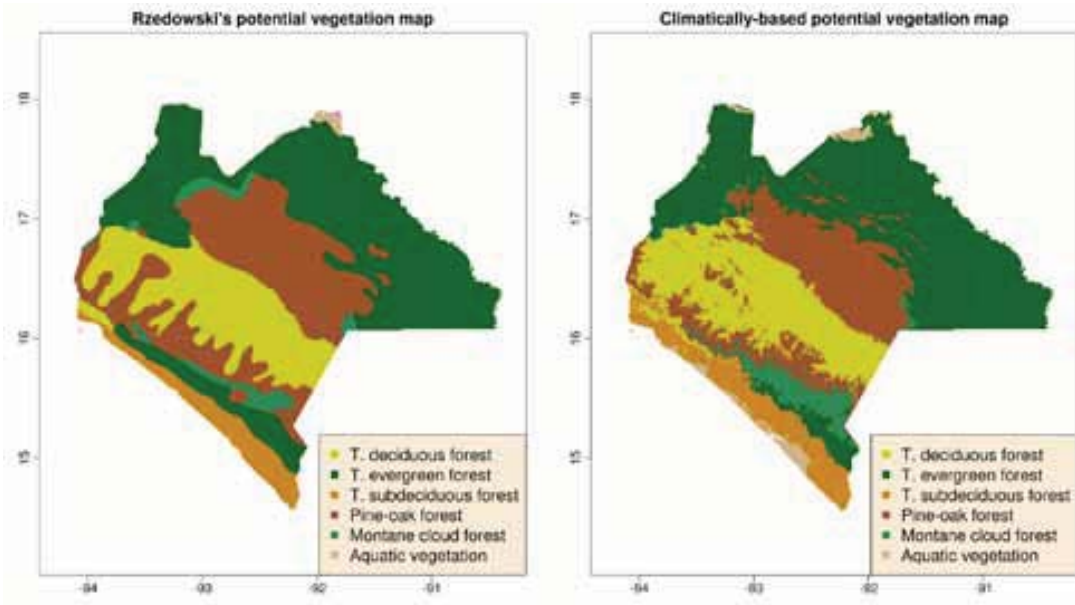
Here, we illustrate the usage of climatic information in the downscaling of coarse-grained potential vegetation maps with reference to the state of Chiapas, a region historically affected by human activity located in Southern Mexico. The aim of this study was to define the original distribution of the area occupied by different tropical vegetation types in the region. At present, one widely recognized source of information on the potential distribution of vegetation types of Southern Mexico is Rzedowski's potential vegetation map (1990) (Olson *et al.*, 2001). This is represented at a scale 1:4,000,000, which is clearly limited as a baseline for the estimation of historical forest loss (Trejo and Dirzo, 2000). In response to this problem, we downscaled Rzedowski's potential vegetation map for Chiapas, from a 1:4,000,000 to 1 km<sup>2</sup> grid resolution, using climatically-based random forests models.

To obtain categorical values for the dependent variable (classes of potential vegetation), we systematically extracted points at one km distance from Rzedowski's potential vegetation map. We did not obtain samples from aquatic vegetation because this vegetation is 'azonal', which means that it is not climatically driven. Once the final potential vegetation map was generated, the distribution of aquatic vegetation was explicitly defined based on a soil map developed for Mexico at a scale 1:1,000,000 (INIFAP and CONABIO, 1995). At each point, we extracted values from 55 climatic variables obtained from the WorldClim site (Hijmans *et al.*, 2005). The dataset consists of 36 grids of monthly mean minimum temperature, maximum temperature and precipitation and a set of 19 bioclimatic variables.

The analysis was performed with 1000 trees. As part of its construction, random forests constructs successive independent trees using a bootstrap sample of the data set, each of which produces a vote (Breiman, 1996). In the end, the set of votes is used to generate a simple majority vote for prediction, or scores that provide basic probability estimates, which may then be used in weighted voting (Fawcett, 2006). We used majority vote prediction rules to generate the downscaled potential vegetation map. We validated the model using 256 inventory plots sampled near the transitions between different vegetation types (as these are the areas that are more inaccurate at the original scale). We used

**Box 2.6 (cont.)**

validation results to get insights into the performance acquired for the different vegetation classes and for overall model. Then we used receiver operating characteristics (ROC) curves to select the decision thresholds (cut-off thresholds based on probability estimates) that encompass the distribution limits for classes with low agreement and maximize overall prediction accuracy (Fawcett, 2006). At the end, we generated a potential vegetation map using the decision thresholds for these classes. Both the original and the climatically downscaled potential vegetation maps are shown in **Fig. 1**. The Kappa Index of Agreement showed an increase in accuracy from 0.40 for Rzedowski's map (95% confidence intervals between 32.5–48.7) to 0.80 for climatically-derived map (95% confidence intervals between 73.9–86.0). Overall accuracy increased from 55.5% to 85.9%. Estimated Kappa for each vegetation class was increased in all cases. Climatically-based random forests models can prove useful to increase the spatial resolution and accuracy of coarse-grained potential vegetation maps in mountainous areas with strong environmental gradients where important climatic variability is obscured at coarse-grained resolution. In conclusion, the proposed method is suitable to generate maps that can be appropriately used as a baseline for the estimation of the historical forest loss.



**Figure 1** Rzedowski's potential vegetation map for Chiapas vs. climatically-based map developed with random forest. (T. = tropical).

production (Rundel, 1999; Balduzzi *et al.*, 1982; Fuentes *et al.*, 1986; Armesto *et al.*, 2007). In addition, successional recovery of forest is usually constrained by continued pressure, water availability, soil erosion, lack of seed banks, disturbance by human-induced fires and limited regeneration capacities of forest species (Balduzzi *et al.*, 1982; Fuentes *et al.*, 1986; Conacher and Sala, 1998; Rundel, 1999; Armesto *et al.*, 2007).

We found that loss of forest extent is partially mitigated by forest re-growth following the abandonment of farmland. In central Chile, we detected forest recovery on about 2.7% of the study area, a rate similar to that documented in other Mediterranean areas (Serra *et al.*, 2008). Forest recovery in central Veracruz may be partially explained as a by-product of comparing data from different types of satellites, but also by declines in agricultural area that may be linked to a reduction in subsidies and increased competition from the US following the NAFTA free-trade accords over the last decade (Pascual and Barbier, 2007). In Oaxaca,

human migration may explain the decrease observed in cultivated land from 1989–1999 (INEGI, 2000). Deforestation in this region was concentrated in a few patches and land-use dynamics are apparently rapid, as traditional crop management involves a 5–10 year continuous cultivation period followed by a 10–15 year fallow period.

## Drivers of forest change

We identified a number of biophysical and socioeconomic variables that were associated with changes in forest extent across our study areas (Table 2.2). Interestingly, the change in forest extent was explained by a unique combination of variables in each study area and the same variable may have either a positive or a negative effect in the different study areas, i.e. in different ecological, socioeconomic and cultural contexts (Boxes 2.2 – 2.7). For the entire time period addressed in this research, the biophysical variables with the strongest effects on change in forest extent were slope, insolation, and distance from remnant forest. Some of these variables drove forest change in opposite directions (loss or gain) in the different study areas. Thus, whereas the probability of an area experiencing forest loss was higher on gentle slopes, in accordance with our hypothesis, insolation showed different impacts in Chiapas (positive correlation) from central Chile and Oaxaca (negative correlation). Proximity to human settlements and farmland decreased the overall probability of deforestation in most study areas – an example of the ‘curtain effect’ – contrary to our hypothesis. Similarly human density was not found to have a major impact on deforestation. Distance from forests or roads had different effects in different study areas (Table 2.3), thereby partially confirming our hypotheses.

**Table 2.3** Synthesis of results of selected multiple models for explaining the effects of various biophysical and socioeconomic variables on the conversion of forest to non-forest land-cover. The significance of the variables is represented by codes, being +++/-- = 0.0001, ++/-- = 0.001 and +/- = 0.01.

Variable	C. Chile	N. Argentina	C. Veracruz	Chiapas	Oaxaca
Elevation				-	+++
Slope	-	---	---	---	
Insolation	---			+++	---
Precipitation		---		-	
Distance to river		+++			
Distance to forest <sup>1</sup>	+++				---
Distance to settlements	+ <sup>2</sup>	++ <sup>3</sup>		+++ <sup>4</sup>	+++
Human density					
Distance to roads	+++	---			+
Distance to agriculture	+++	++			+++
Distance to pasture			--		

1 Distance within forest edge for the study area in central Chile

2 Cities >20,000 inhabitants

3 Towns <20,000 inhabitants

4 Access to towns >5,000 inhabitants

## **Box 2.7** Different sets of drivers across study regions

In central Chile, the multivariate logistic regression model for forest-no forest revealed that the probability of an area experiencing forest loss was highly significant ( $p < 0.001$ ) and positively related to the distance from the nearest forest edge for the four study periods, i.e. deforestation progresses from within forest fragments towards the edge, and produces treeless gaps within the forest. This variable had the strongest partial deviance of the four resulting models, accounting for at least half of the deviance explained by the final models achieved in the stepwise selection procedure. For the whole study period, the main explanatory variables of deforestation after distance from the nearest edge were distance from roads and distance from agriculture, all of which were positively correlated with the probability of deforestation, while insolation and slope were both negatively correlated and less relevant in terms of explained variance.

In northern Argentina, the probability of forest loss was highly related ( $p < 0.001$ ) to slope (negative correlation) and distance from rivers (positive correlation) in all study periods. Other significant explanatory variables were distance from roads and mean annual precipitation, showing that deforestation started close to roads and with relatively high precipitation (for dry forests), and moved away from those ideal conditions to current areas further away from roads and with lower precipitation. Distance from the agriculture frontier and from villages showed a significant positive effect, with higher deforestation rates in areas of higher accessibility and human presence.

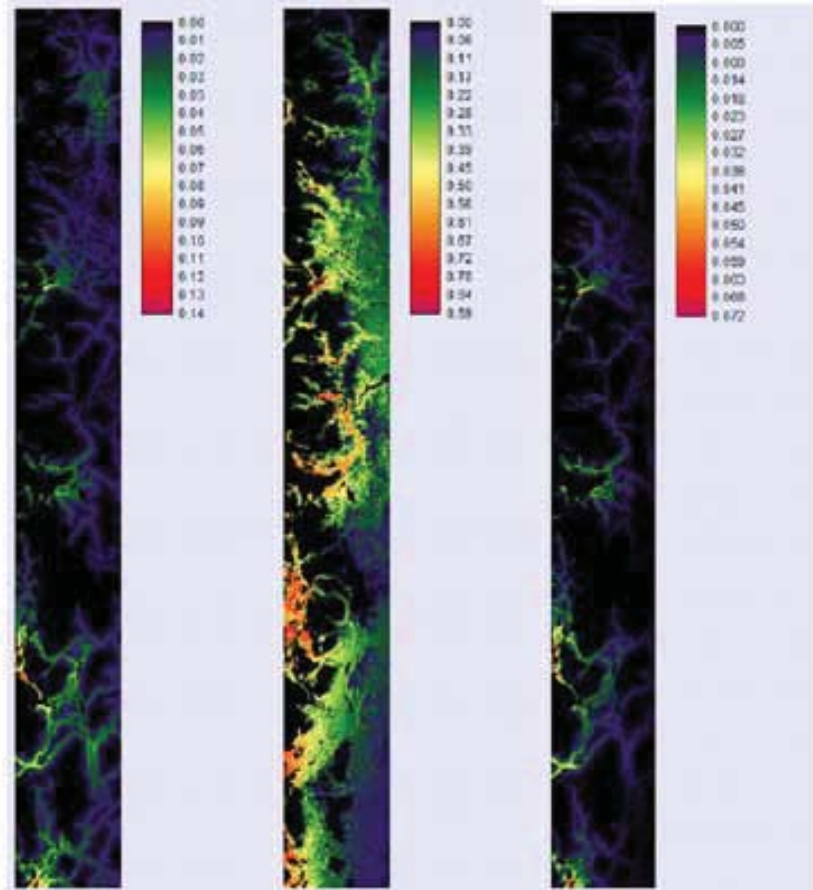
In central Veracruz, univariate logistic regression models indicated that only four variables were negatively correlated with the probability of forest transformation. They included, in decreasing order of importance, slope, distance from pastures, distance from irrigation infrastructure, and aspect. When these same variables were incorporated into a multivariate logistic regression, the resulting model was highly significant according to the analysis of deviance, although the percentage of deviance explained was relatively low (19.4%). Slope and distance from pasture were found to be significant at the  $p < 0.001$  level, whereas the importance of distance from irrigation infrastructure ( $p = 0.063$ ) and aspect ( $p = 0.09$ ) declined.

In Chiapas, a GAM model which included elevation and annual rainfall explained the largest proportion of the deviance (12.2%). A model without rainfall explained 11.6% of the deviance, and slope became the variable most strongly associated with the probability that a pixel remained forested. After slope, radiation was the second most important variable both when taken alone and within a multivariate model. Access to large towns (>5000 inhabitants) had greater explanatory power as measured by deviance and partial deviance than access to small villages (>100 inhabitants). The probability that forest has been lost was found to be associated with high values of insolation during winter months, gentle slopes, accessibility to principal markets, low elevations, and low annual precipitation.

In Oaxaca, the probability of forest loss was highly significant and positively related to distance from crop fields and distance from villages in the four study periods. For the entire period (1979–2005), the main explanatory variables for deforestation with significant positive correlations were elevation and distance from agriculture, from villages and from forest; those with significant negative correlation were insolation and distance from forest.

Dryland forest areas in southern Argentina are undergoing a small but significant change to plantations with exotic conifers. Afforestation with exotic pines during the 1973–2003 period tended to occur at significantly ( $p < 0.05$ ) shorter distances from roads, urban areas and towns >1000 inhabitants. Distance from rivers or lakes seems to have been a poor predictor of afforestation. Contrary to expectation, it tended to occur at longer distances from small towns. Afforestation tended to occur at low elevations and on gentle slopes and at areas with mean annual precipitation higher than the average of the study area. This model was useful to detect possible land-use conflicts with the afforestation process. For example, it has been

suggested that afforestation with exotic conifers has taken place at the expense of land potentially suitable for passive or active restoration of dryland forests of the native conifer *Austrocedrus chilensis*. If we compare potential for transition towards afforestation with habitat suitability for this species, we obtain the land-use conflict map presented in Fig. 2.4.



**Figure 2.4** Application of the multivariate logistic model of afforestation to identify areas of potential dryland forest restoration in conflict with other land-use types in southern Argentina. *Left:* Map illustrating potential for afforestation with exotic pine species derived from this study. *Centre:* Habitat suitability model for *Austrocedrus chilensis*, the main native dryland forest species in the region. *Right:* Areas of conflict that are highly suitable for restoration of native dryland forest (or already have dryland forest) but are at risk of being converted to exotic pine plantations. Higher values indicate higher risk of conversion.

The differentiated analysis of drivers of deforestation indicated that the trend towards a reduction in natural vegetation cover was determined by a variety of biophysical and socio-economic contexts that resulted in different patterns of land-use types. For the entire time period addressed in this research, the biophysical variables with strongest effects on change in forest extent were slope (higher deforestation in flat areas), insolation, and distance from forest remnants. Slope and associated topographical barriers are fairly typical drivers identified in studies of tropical forest transformation (Geist and Lambin, 2002). In central Chiapas, fire is used to clear forests and to prevent woody re-growth in cattle pastures; therefore, slopes that receive more insolation in the dry season are more likely to be permanently cleared of woody vegetation than shaded slopes. Moreover, Echeverría *et al.* (Chapter 3) have

shown that fire frequency and extent are increasing in these forest landscapes, making them more vulnerable to desertification.

Climate change, particularly rainfall patterns, may be also linked to deforestation (Grau *et al.*, 2005). In recent years, there has been an increase in rainfall in northwestern Argentina (Villalba, 1995), leading to expansion of the agricultural frontier and contributing significantly to the rapid increase in deforestation in the region. This has been accompanied by technological improvements (e.g. genetically modified soya) and high international demand that have raised product prices.

In central Chile, forest loss occurred with higher probability inside forest stands than at the border. As a consequence, our analyses also detected a higher probability of deforestation at larger distances from roads and agricultural fields. The same pattern has been observed in other studies (Ochoa-Gaona and González-Espinosa, 2000) and reveals hidden pressures from cattle grazing and illegal logging activities such as firewood collection and charcoal production (Armesto *et al.*, 2007). Such hidden pressures are not rare in Latin American countries (Callieri, 1996; Aubad *et al.*, 2008) where the rural population often depends on firewood for household consumption as well as the illegal production of charcoal for income generation. In central Veracruz, clearly, the rapid expansion of irrigated agriculture and cattle ranching increased pressure on native forests. Human activity-related factors affect forest fragment accessibility, as has been reported in other studies (Fujisaka *et al.*, 1996; Wassenaar *et al.*, 2007). Settlers converted land to pasture not only to raise cattle, but also to establish unpaved roads and collect firewood. After the forest fragments near the pastures are degraded, land conversion to agriculture or other pasture is more probable. In southern Argentina, deforestation is presumably produced by the occurrence of natural and anthropogenic fires, which in many cases do not regenerate back into forests and remain as stable grasslands or shrublands (Mermoz *et al.*, 2005). However, dryland forest areas in this region are undergoing a small but steady change to plantations with exotic conifers. This trend, in concert with other threats such as anthropogenic fires, livestock grazing and the introduction of exotic herbivores such as hares and rabbits are factors that hinder the restoration of dryland forest, which has been fragmented over centuries by native populations and European colonists (Veblen and Lorenz, 1988). Models of land-use/land-cover change can aid the identification of target areas of low conflict for a more rational planning of restoration efforts.

## **Implications for landscape planning and management**

Human interactions with ecosystems are inherently dynamic and complex, and any categorization of these is an oversimplification. However, there is little hope of understanding these interactions without such simplifications (Ellis and Ramankutty, 2008). Working in multiple regions within Latin America enabled us to identify general trends at the regional scale that might be useful for landscape planning and serve as a basis for analyzing proximate drivers of land-cover change. However, the disadvantage of this approach is that it is more difficult to identify and assess patterns and process of land-use change at local scales in the real world (e.g. an individual field). Nevertheless, informal interviews that were conducted alongside the field surveys to establish classifications or to ascertain accurate assessments in most study areas provided an important complementary source of information to interpret the detected changes at the regional scale.

Natural vegetation loss and degradation reduce precipitation infiltration and runoff regulation, which promotes soil erosion, landslides and avalanches, and has a negative

impact on ground water recharge (Conacher and Sala, 1998; Millennium Ecosystem Assessment, 2005b). In addition, vegetation cover is tightly associated with water balances within watersheds, biodiversity conservation, and regional climate regulation (Maass *et al.*, 2005; Feddema *et al.*, 2005; Foley *et al.*, 2005; Pielke, 2005). Land-use decisions therefore have consequences for the structure and function of ecosystems and affect provision of environmental goods and services; these decisions also affect humans in ways that go beyond the immediate land-use situation (Turner *et al.*, 2007). The continuous degradation of vegetation cover could have a strong impact on human livelihoods and well-being in the studied dryland landscapes, as there are increasing water demands for agriculture (Cai *et al.*, 2008) and human consumption owing to large population increases.

Environmental problems such as degradation, loss of biodiversity and decreases in productivity accumulate over the long term and have non-linear effects at regional to global scales (DeFries *et al.*, 2004; Foley *et al.*, 2005). Consequently, strategies for adapted land use, including the optimization of the spatial configuration of uses and restoration of the natural vegetation cover in critical areas should be developed quickly. Strategies should go beyond preservation within protected areas and logging restrictions along rivers and streams (Turner *et al.*, 2007). For instance, Rey Benayas *et al.* (2008) proposed the 'woodland-islet in agricultural seas' model to conciliate agricultural production and conservation or restoration of native woodlands. Closer monitoring is needed of livestock to establish guidelines for an adapted carrying capacity, as cattle also graze in forests. The repercussions of unsustainable firewood extraction and charcoal production have hardly been quantified in many regions, but we know that they impact strongly on forest conservation (see Chapter 6).

Land-use planning at the regional scale provides a unique opportunity for the establishment of general strategies that may, on one hand, accept or even promote deforestation at particular selected areas, and on the other hand maintain large forested areas suitable for sustainable timber and non-timber forest uses, and probably to a lesser extent, areas for conservation purposes. In northern Argentina, a land-use planning policy has been implemented over 10 million hectares of dry forests, zoning different land uses, from deforestation to conservation. Most of the forest corresponds to an intermediate category, theoretically oriented towards forest uses compatible with its own maintenance in the long run. In practice, most forest is heavily degraded and major efforts should be made to find economic incentives for the local inhabitants to reverse the degradation process and provide value to the remaining forests (see Chapter 10).

Apart from the need for land-use planning, restoration and rehabilitation are important issues in drylands (Le Houerou, 2000; Vallejo *et al.*, 2006). Long-term land-use intensification may represent unique cultural challenges for restoration efforts owing to the long history of human activity, the period of time during which dryland forest has been reduced and degraded, and generations of inhabitants have grown accustomed to its absence in the studied regions (Piegay *et al.*, 2005; Hobbs, 2009). In Chile, Holmgren and Scheffer (2001) postulated that there might be a window of opportunity for passive restoration through the exclusion of herbivores in El Niño Southern Oscillation (ENSO) years owing to higher water availability; this strategy could also be applied in southern Argentina given similar ENSO-climate connections. It could be especially interesting to use this strategy to establish buffer zones and corridors between remaining old growth forest, which were detected in this study as stable forest areas. Also, forms of adaptive and multifunctional land use such as mixed agroforestry systems should be encouraged as an alternative to monoculture cropping and crop pasture rotations (Ovalle *et al.*, 1996; Aronson *et al.*, 1998).



## **Conclusion**

The research described here has provided quantitative estimates of forest extent and characterized the changes in land-cover in a wide variety of dryland landscapes under contrasting ecological, socioeconomic, and cultural scenarios. In addition, research examined the dynamics and drivers of forest loss that has taken place over the last ca. 30 years. We concluded that land-use intensification and limited natural regeneration continue to threaten dryland forest cover in many regions of Latin America, but that deforestation rates have diminished in the recent past compared to trends in the early part of the twentieth century, in accordance with global trends. The probability of an area experiencing forest loss was found to be higher on gentle slopes, and surprisingly, proximity to human settlements and farmland decreased the probability of deforestation in most study areas. Such analyses can help identify those areas that supported native forest in the past, and might therefore be considered as candidates for restoration. In addition, analysis of the factors responsible for forest loss and degradation can inform the development of restoration strategies and plans, by identifying those threatening processes that need to be addressed if restoration actions are to be successful.

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