3 ASSESSING FRAGMENTATION AND DEGRADATION OF DRYLAND FOREST ECOSYSTEMS

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Introduction

Spatial patterns of forest cover can be understood as the spatial arrangement or configuration of forested ecosystems across a landscape (Forman and Godron, 1986). The importance of studying spatial patterns of forest cover is now widely appreciated, owing to the complex link between pattern and process in a landscape (Nagendra et al., 2004), and the widely documented effects of habitat fragmentation on biodiversity. As a result, diverse studies have sought to develop measures of landscape pattern that may be used to monitor changes in forest cover (Sano et al., 2009; Shuangcheng et al., 2009; Zeng and Wu, 2005).

According to the driving factors that operate in a given landscape, spatial pattern can present a variety of different behaviours over time. For instance, loss and fragmentation of forest cover are among the most important transformations of landscape configuration occurring in many parts of the world (Carvalho et al., 2009; Fialkowski and Bitner, 2008). On the other hand, pattern change associated with forest recovery or regeneration may lead to an increase of forest cover and connectivity (Baptista 2010; Box 3.1).

Box 3.1 Landscape features associated with the passive recovery of Mediterranean sclerophyllous woodlands of central Chile

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Although Mediterranean ecosystems are considered global hotspots of biodiversity and priority targets for conservation (Myers et al., 2000), they are among the most severely degraded and fragmented ecosystems in the world. In central Chile, land-cover of Mediterranean sclerophyllous woodlands (Chilean matorral) has been significantly reduced and transformed by a combination of human activities including logging, firewood extraction, vegetation burning, agriculture, livestock grazing, and the spread of exotic species of herbivores (Fuentes and Hajek, 1979; Holmgren, 2002; see Chapter 2). Ecosystems that have been highly degraded and extirpated from large areas, such as Chilean Mediterranean forests, are difficult and expensive to restore, especially because of the extremely dry and long summer period and strong impact of herbivory. Both factors, in addition to recurrent fire, can stop or retard successional processes (Fuentes et al., 1984). Frequently, severely degraded dryland ecosystems cannot be returned to their pre-disturbance condition without expensive management. The less costly strategies to restore vegetation cover in these ecosystems is to combine the passive regeneration of relatively less impacted areas, resulting from relatively slow natural processes, with active restoration activities that stimulate vegetation change from early successional stages to more mature and diverse forest.
Box 3.1 (cont.)

We assessed the regeneration potential of sclerophyllous woodlands of central Chile (33° S) over 50 years at three sites in the foothills of the Andes and two sites in the Coastal Range, and related the rates of vegetation change to specific landscapes features. Each study site (Fig. 1) was a mosaic of sclerophyllous forest and open pastures, with an average 40% of woodland cover and an extension of 700 ha on average (range: 631–911 ha) and had not been burned for at least two decades (1985–2008). Vegetation change was determined by comparison of aerial photographs taken in 1955 and 2007 over a regular grid of 250 m of points using standard supervised classification methods. We considered as evidence of woodland regeneration (1) a change in land-cover for a given point in the grid from bare soil or artificial grassland to forest cover. Persistence of the open cover condition was considered as a lack of forest regeneration (0). Any other observed changes in the vegetation or the maintenance of forest cover were excluded from the analyses. We related the recovery of forest cover to topographic variables (slope, orientation, altitude and exposure to solar radiation), as well as to spatial location of the regenerating patch (distance to the closest forest patch present in 1955, and distance to the nearest ravine). We used spatial regression models to control for spatial autocorrelation among sampling points.

We found an average rate of increase in land-cover of sclerophyllous forest from 0.4–1.0 ha/year. The probability of recovery of forest cover increased significantly at shorter distances from remnant (1955) forest patches, especially on south-facing slopes. This effect may be related to fact that patches can be a source of propagules but their environmental conditions may also facilitate tree seed germination and seedling survival (Fuentes et al., 1984, 1986; Holmgren et al., 2000). The spatial regression models also suggest that regeneration occurs in patches (at a 250 m scale), which could be related to local differences in grazing pressure, resource availability (nutrients and water) and micro-climatic conditions (temperature and air relative moisture).

Our work shows that Chilean sclerophyllous forest, which is considered strongly resistant to passive recovery from severe disturbance, can grow back in unburned sites under certain conditions. The proximity to existing forest patches or seed sources, slope aspect, and the aggregated patch structure of the vegetation are key features to be considered in the design of successful long-term restoration strategies to promote the passive restoration of Mediterranean sclerophyllous woodlands. The removal of herbivores, if possible, could accelerate the passive recovery of woodland vegetation cover (see also Chapter 8).

Figure 1    Location of the five study sites in central Chile (Cachagua, Los Molles, San Carlos de Apoquindo, Pirque). The aerial images show vegetation changes between 1955 and 2007 in Río Clarillo. Darker areas represent evergreen forest.
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Central Valley of Chiapas, Mexico; tropical deciduous forest. Photo: R. Vaca.

Deforestation of seasonal dry premontane forest in northwestern Argentina. Photo: L. Malizia
Progressive deforestation typically results in an increase in the spatial heterogeneity, fragmentation, and edge characteristics of a forested landscape (Trani and Giles, 1999). In particular, fragmentation refers to the division of spatially continuous forest areas into isolated patches, which are separated by some other type of land-cover (such as agricultural land), commonly referred to as the landscape matrix (Forman and Godron, 1986). At the patch level, fragmentation causes an increase in patch isolation and edge, and a reduction of patch size (Echeverría et al., 2006). In turn, this can increase the isolation of populations of individual species (Echeverría et al., 2007), and can reduce population viability through its effects on key ecological processes such as dispersal, migration and gene flow (Giriraj et al., 2010; Vergara and Armesto, 2009). As a result, forest fragmentation is now considered to be one of the principal causes of biodiversity loss (Baillie et al., 2004). As forest loss takes place in a landscape, certain changes in the spatial configuration of the landscape can be observed (Cayuela et al., 2006; Geri et al., 2009). The analysis of the spatial attributes through landscape indices is a suitable approach to demonstrate the process of forest fragmentation at the landscape level (Zeng and Wu, 2005). Additionally, information on landscape structure can be used to inform forest management (Sano et al., 2009).

Dryland systems are recognized as being of high biodiversity value, while representing the largest terrestrial biome on the planet (MEA, 2005; Schimel, 2010). Throughout the areas where they occur, dryland forests have been rapidly degrading and declining owing to anthropogenic disturbance (Hill et al., 2008; Ravi et al., 2010; Reynolds et al., 2007). Loss of dryland forests has had a significant impact on carbon sequestration and temperature at the global scale (Rotenberg and Yakir, 2010). In Latin America, this ecosystem has been associated with human poverty, unhealthy living conditions and environmental degradation (Altieri and Masera, 1993). Management for the conservation and sustainable use of dryland forests should consider restoration approaches and habitat modification (McIntyre and Hobbs, 1999) with the aim of enhancing both biodiversity and human livelihoods. Few studies on the spatial pattern of dryland landscapes have been performed to examine the effects of human activity on dryland forests (Wang et al., 2010), particularly in the context of their restoration.

In this chapter we present the results of research that assessed the trends in landscape patterns of forest cover in seven dryland study areas in Mexico, Argentina and Chile, by analyzing the dynamics of selected landscape metrics over the last four decades. Through a comparative analysis of these study areas, we identified both the high variability of landscape metrics and common trends in the spatial patterns of dryland forest. The aim of this research was to inform the development of plans for forest landscape restoration, one of the objectives of which is to restore connectivity in forest areas that have been fragmented (Box 3.2). Examination of the patterns and processes influencing forest fragmentation are therefore of direct relevance to the development of restoration approaches to be implemented at the landscape scale. The study landscapes included Veracruz, Oaxaca and Chiapas in México; Salta in northern Argentina and Bariloche in southern Argentina; and central Chile (Fig 3.1). A set of Landsat satellite images was classified and analyzed in each study area (see Chapter 2). Most of the study intervals spanned the past four decades, except for Chiapas where the period from 1990 to 2005 was analyzed.
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Figure 3.1  Location of the study areas in Latin America.

Figure 3.2  Study periods used for the fragmentation analysis in each study area.
Box 3.2  Landscape connectivity in the highly fragmented drylands of the Central Valley of Chiapas

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

The current pattern of forest cover observed in the Central Valley of Chiapas is the result of historic deforestation. We found that 68% of the original hypothesized area of dry forest (as defined by Olson et al., 2001) was lost by 1990. The remaining forest (32% of the original putative area) is either retained by local landowners for its utility as a source of fuelwood and timber, or located in nature reserves or sites with steep slopes and low accessibility. Most of the forest in this region is highly fragmented, and only 19% is found in core areas, i.e. forest with a minimum distance of 110 m from the nearest patch edge. This landscape, dominated by human land use, presents a significant challenge for maintaining and conserving biodiversity.

Although most of the original forest cover had already been lost, the forest has not been completely cleared and replaced with an inhospitable matrix, as has occurred in other agricultural croplands or suburban environments of the world. The spatial concept of fragmentation in this case does not necessarily imply that habitat remnants are isolated by areas that function as hostile environments to the organisms within the remnants (Cayuela, 2009). The agricultural landscape still retains large isolated trees, woodlots, scattered groups of trees, secondary regrowth, hedgerows and living fences, amongst forest and shrubland patches of varying size, disturbance and management history. Together, they provide the habitats upon which the conservation of much of the flora and fauna in developed landscapes ultimately depends (Bennett, 1998; 2003). Even though regenerating or degraded forests and isolated trees may not provide all of the resources that a particular species need to survive, they may pose little resistance to the movement of many animals between patches and protected areas of forest where these resources are available (Bennett, 1998; 2003). In this context, an important priority for biodiversity conservation is to maintain a mosaic of semi-natural connected habitats within the agricultural land.

To investigate this issue, we measured dry forest connectivity in the study area (Fig. 1a) based on two different approaches, and we identified barriers to movement and priority actions for the region. The first approach focused on forest specialist species, i.e. species that have strict forest requirements. These species therefore require core areas for their survival over the long term. We focused on core areas of continuous forest cover larger than 5 ha. We developed a connectivity analysis based on distance matrices. Cores were considered neighbours if the smallest distance between their edges was less than 4 km (Figure 1b). The second approach focused on species that are less restricted in their forest habitat requirements, and can use isolated trees or small woodlots as well as core areas, and disperse easily through the matrix (e.g. some birds, insects, and many pioneer plant species). For these species, a highly fragmented landscape becomes more permeable to dispersion. In this case, we developed a connectivity analysis buffering away from any pixel classified as tree cover, using different buffer distances (100 m, 200 m, 300 m, 500 m, 1000 m, 5000 m, etc.). The map developed through this analysis shows the distance zones (proportion of area) between pixels classified as tree cover (Figure 1c). This analysis thus allowed the recognition of areas of decreasing permeability to movement.
Box 3.2 (cont.)

**Figure 1a** Study area, the Central Valley of Chiapas: forested areas are represented by dark green and the non-forested matrix by tan.

**Figure 1b** Connectivity analysis based on distance matrices: black lines represent Euclidean distances of less than 4 km between the edges of cores with areas higher than 5 ha.
Box 3.2 (cont.)

Barriers to movement tended to coincide spatially using both approaches. Results suggest that core areas were generally not well connected, especially in the centre of the study area. Nevertheless, isolated trees and small patches may enhance connectivity considerably for mobile organisms. The distance between any form of tree cover was generally below 200 m. The lowest connectivity was found in the area around El Parral (pointed out in Fig. 1a). But even in this area, trees were still present (Fig. 2). Biodiversity conservation can be achieved by maintaining the diffuse mosaic of forest, open woodland and scattered trees, but also through restoration of habitats focused both on linking core areas and increasing permeability. The forest can be conserved by working with landholders in order to minimize human impacts in the remaining forest patches, many of which are greatly disturbed and degraded as a result of livestock grazing. Future actions for increasing connectivity and permeability should target the restoration of degraded pastures, the development of fuelwood plantations, and the expansion of living fences, shade and forage trees within the landscape. Finally, further actions should be focused on protecting and managing major links between conservation reserves to assist their long term viability.

Figure 1c Connectivity analysis based on buffers of different distances, from any pixel classified as tree cover. Forested pixels are represented by dark green, and buffers of increasing distance are represented by a colour gradient: light green (buffer areas from 0 to 200 m), orange (buffer areas from 200 to 500 m), and red (buffer areas from 500 to 5000 m). This colour gradient represents the proportion of area between pixels classified as tree cover, pointing out areas of decreasing permeability to movement.

Figure 2 Deforested drylands near El Parral in the Central Valley of Chiapas, Mexico.
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Cropland in dry forest areas of Chile. Photo: C. Echeverria

Dry forest in central Veracruz, Mexico. Photo: C. Alvarez
Techniques to quantify spatial patterns of forest cover

Analysis of forest fragmentation was conducted using the following set of selected landscape metrics: (a) patch area (ha), (b) proximity index, (c) patch density (n/100 ha), (d) total edge length (km), and (e) largest patch index (LPI, %). All of these metrics reflect the different effects of fragmentation on the spatial attribute of forest patches. Index proximity was calculated for a radius of 1 km and core area for an edge depth of 50 m. In addition, we estimated aggregation index and adjacency index between forest cover and the major land-cover types. It is expected that the aggregation of forest patches decreases as a result of fragmentation and increases with forest contiguity. Similarly, the adjacency between native forest and human-induced land-cover types should increase with changes in the matrix.

A minimum mapping unit of greater than 5 pixels was used for the spatial analyses. This enabled differences in data quality produced by the resampling of the MSS images to be minimized. Map preparation was performed using ARC MAP (version 3.3; ESRI, 2009). Landscape metrics were computed by FRAGSTATS (version 3) (Mcgarigal et al., 2002) to compare the spatial patterns of forest cover for each time interval and study area.

Mapping spatial patterns of forest cover

Maps of forest cover based on patch size were generated for each study area and for each study year (Fig. 3.3). Most of these maps provide evidence of typical patterns of anthropogenic landscape change. The patterns are comparable to those observed in many other parts of the world (Abdullah and Nakagoshi, 2006; Wang et al., 2010). Deforestation and fragmentation of dryland forests have occurred in most of the study areas, except in Bariloche where some forest fragments moved to upper classes of size during the study interval (Fig. 3.3c), and in Chiapas, where forest fragments did not appear to change in size over time (Fig. 3.3b). Maps of forest fragmentation showed a considerable increase in the number of smaller patches over time in Xalapa, Oaxaca, central Chile and Salta (Figs. 3.3a, d, e and f respectively).

Burned stand of *Austrocedrus chilensis* in Southern Argentina. Photo: J. Birch
Figure 3.3  Temporal variation in patch size for the different study areas: (a) central Chile, (b) Chiapas, Mexico, (c) Bariloche, Argentina, (d) Salta, Argentina, (e) Veracruz, Mexico, (f) Oaxaca, Mexico. Larger patch sizes in green, smaller patch sizes in red, intermediate patch sizes in orange.

Figure 3a  Central Chile
Figure 3b  Chiapas, Mexico

Figure 3c  Bariloche, Argentina
Figure 3d Salta, Argentina
Figure 3e  Veracruz, Mexico
Figure 3f  Oaxaca, Mexico
Analysis of spatial patterns of dryland forests

Most of the study areas exhibited a decline in patch size between the earliest and the most recent maps. In particular, Veracruz, Oaxaca, central Chile and Salta showed a decline in patch size, while Chiapas remained practically constant and Bariloche exhibited an increase in this variable (Table 3.1). In Oaxaca and central Chile, the total edge length of forest patches increased and then declined over time (Table 3.1). In Veracruz and Salta there was an increase in the total of edges. In contrast, Bariloche was the only study area that exhibited a permanent decline in the number of fragment edges, whereas Chiapas did not demonstrate changes in this variable over time. With respect to the core areas of forest fragments, all of the study areas exhibited a decline in this index through time, except for Bariloche, which showed an increase (Table 3.1). The greatest declines occurred in central Chile and in Veracruz, where 66% and 51% of the core area was lost during the study periods respectively. In contrast, Chiapas did not present a substantial change in this index (1.3%), while in Bariloche the core area increased 16% (Table 3.1). Index of proximity (which provides a measure of the degree of isolation) decreased in Chiapas, central Chile and Salta (Table 3.1). In Veracruz and Oaxaca, this index varied during the study period, without providing a clear trend. In Bariloche this index presented an increase between 1973 and 1997 and then it declined during the last time interval.

These trends in landscape indices were associated with variation in patch density (Fig. 3.4). Owing to the fact that the number of patches may increase as a result of the creation of new patches by fragmentation, a further decline can be observed either by the loss of the new forest patches or the union of patches as a result of forest regeneration. This trend enabled different stages in the spatial dynamics of forest to be identified. In Veracruz and Salta, a gradual increment in patch density (Fig. 3.4) and edge length, and a decline in patch size and core areas (Table 3.1) characterized a landscape affected by progressive fragmentation during the study periods.

Oaxaca is the only study area where the patch density and edge length were curvilinear, with metrics changing direction at the half-way point of the study period (Fig. 3.4). This reflects a rapid division of forest patches that were later eliminated by high rates of deforestation. In central Chile, the number of patches gradually decreased owing to the conversion of forest patches (Fig. 3.4). This pattern was associated with a decrease in core area and an increase in patch isolation, and with a continuous loss of forest fragments over time (Table 3.1). In contrast to this situation, in Bariloche the increase in patch size, core area and proximity index and a decline in patch density and edge length showed that the forest cover was recovering, showing an opposite trend to forest fragmentation (Fig. 3.4 and Table 3.1). In Chiapas, the slight decline in patch density (Fig. 3.4), and the almost constant values of patch size, edge length and core area, revealed a low level of forest fragmentation in this landscape and the stabilization of forest cover (Table 3.1).

During the study periods, the largest forest patch occupied no more than 2% of the whole landscape in each of Chiapas, central Chile and Bariloche. On the other hand, in Salta this index reached a higher value, ranging from 52% to 32% between 1977 and 2006. In Oaxaca values varied slightly from 24% to 23%, and in Veracruz, from 6% to 2%.
Table 3.1 Landscape metrics estimated for the six study areas.

**Veracruz, Mexico**

<table>
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<tr>
<th>Landscape indices</th>
<th>1973</th>
<th>1990</th>
<th>1999</th>
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<tr>
<td>Mean patch size (ha)</td>
<td>139.7</td>
<td>73.9</td>
<td>27.9</td>
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<td>Total edge length</td>
<td>3,091,320</td>
<td>6,334,110</td>
<td>6,345,420</td>
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<tr>
<td>Total core area (ha)</td>
<td>44,160.84</td>
<td>41,404.14</td>
<td>21,771.7</td>
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<tr>
<td>Mean proximity</td>
<td>1,235.1</td>
<td>16,712.5</td>
<td>1,229.35</td>
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**Oaxaca, Mexico**

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<th>1979</th>
<th>1989</th>
<th>1999</th>
<th>2005</th>
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<tr>
<td>Mean patch size (ha)</td>
<td>99.9</td>
<td>22.9</td>
<td>41.2</td>
<td>46.7</td>
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<tr>
<td>Total edge length</td>
<td>64,069.8</td>
<td>105,900.9</td>
<td>106,688.9</td>
<td>89,516.0</td>
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<tr>
<td>Total core area (ha)</td>
<td>514,323.3</td>
<td>2,246,893.7</td>
<td>386,258.9</td>
<td>428,649.66</td>
</tr>
<tr>
<td>Mean proximity</td>
<td>332,727.6</td>
<td>28,168.4</td>
<td>366,210.5</td>
<td>388,341.7</td>
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**Chiapas, Mexico**

<table>
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<th>Landscape indices</th>
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<th>2005</th>
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<tr>
<td>Mean patch size (ha)</td>
<td>13.1</td>
<td>14.1</td>
<td>14.4</td>
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<tr>
<td>Total edge length</td>
<td>113,509.0</td>
<td>110,960.3</td>
<td>111,196.6</td>
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<td>Total core area (ha)</td>
<td>277,821.0</td>
<td>275,821.0</td>
<td>274,287.0</td>
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<tr>
<td>Mean proximity</td>
<td>4,432.03</td>
<td>4,174.92</td>
<td>4,164.15</td>
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**Central Chile**

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<td>Mean patch size (ha)</td>
<td>8.8</td>
<td>6.3</td>
<td>6.2</td>
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<td>Total edge length</td>
<td>44,400.1</td>
<td>49,857.8</td>
<td>50,768.7</td>
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<td>Total core area (ha)</td>
<td>76,901.2</td>
<td>29,922.8</td>
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<td>26,149.2</td>
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<td>Mean proximity</td>
<td>1,028.0</td>
<td>454.9</td>
<td>380.9</td>
<td>427.1</td>
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**Salta, Argentina**

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<td>Mean patch size (ha)</td>
<td>1,074.7</td>
<td>757.9</td>
<td>528.6</td>
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<td>Total edge length</td>
<td>8,540.1</td>
<td>15,826.2</td>
<td>15,455.0</td>
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<tr>
<td>Total core area (ha)</td>
<td>682,693.0</td>
<td>614,457.0</td>
<td>581,090.0</td>
<td>506,464.0</td>
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<tr>
<td>Mean proximity</td>
<td>299,131.0</td>
<td>522,803.0</td>
<td>217,447.0</td>
<td>174,601.0</td>
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**Bariloche, Argentina**

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<tbody>
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<td>Mean patch size (ha)</td>
<td>8.85</td>
<td>14.3</td>
<td>12.2</td>
<td>13.9</td>
</tr>
<tr>
<td>Total edge length</td>
<td>79,359.68</td>
<td>52,586.80</td>
<td>62,149.86</td>
<td>52,021.08</td>
</tr>
<tr>
<td>Total core area (ha)</td>
<td>115,080</td>
<td>135,654</td>
<td>145,918</td>
<td>132,901</td>
</tr>
<tr>
<td>Mean proximity</td>
<td>1854</td>
<td>2226</td>
<td>2336</td>
<td>2059</td>
</tr>
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</table>
As forest loss continues, it is expected that the largest patch index (LPI) will decline owing to a division of large patches by fragmentation (Trani and Giles, 1999). By graphing the forest loss versus LPI for the study areas, it was observed that in Oaxaca, Veracruz and Salta (Fig. 3.5) a continuous fragmentation has led to a division of large forest patches, causing a decline in the LPI. However, in central Chile and Bariloche, there was a slight increase in LPI as forest loss increased (Fig. 3.6). This opposite trend was the result of the union of large forest fragments despite the loss of others (Fig. 3.3). In Chiapas the LPI did not show variation, owing to the fact that forest area remained almost constant during the study period (Fig. 3.6).
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Figure 3.5  Relationships between forest loss and largest patch index in study areas where the largest patch represents more than 4% of the area of the landscape: (a) Salta, Argentina, (b) Veracruz, Mexico, (c) Oaxaca, Mexico.

Figure 3.6  Relationship between forest loss and largest patch index in study areas where the largest patch represents less than 4% of the area of the landscape. Different scales in the x axis were used for clarity: (a) central Chile, (b) Chiapas, Mexico, (c) Bariloche, Argentina.
Results also showed different rates of forest disaggregation over time, illustrated by the degree of forest loss and fragmentation (Fig. 3.7 and 3.8). The greatest decline in forest aggregation was observed in central Chile, where this index decreased from 82% to 65% across the study period, most rapidly during the first time interval (Fig. 3.7). However, in the earliest study years, Salta and Veracruz exhibited the highest levels of forest aggregation or spatial integrity, with 99% and 96% respectively (Fig. 3.7 and 3.8). In central Chile the disaggregation of forest cover was accompanied by a loss of forest patches rather than by a division of forest patches, as demonstrated by values of patch density (Fig. 3.4). On the other hand, in Veracruz and Salta the number of patches increased (Fig. 3.4) while patch size declined (Table 3.1), reflecting a gradual decline in the level of forest aggregation or an increase in forest fragmentation. Chiapas remained constant with 85% forest aggregation, indicating no change in spatial patterns (Fig. 3.7).

Oaxaca was the only study area that exhibited a decline and further increase in the aggregation index over time (Fig. 3.12). By comparing this result with the values generated for the other metrics, it can be observed that during the first time interval, forest cover was disaggregated by the division of a large patch, which resulted in an increase in the number of patches (Fig. 3.8). Between 1989 and 2005, the forest became more aggregated, increasing the patch size (Table 3.1). In Bariloche, forest cover showed a gradual increase in aggregation (Fig. 3.12), indicating the recovery of new patches and an increase in patch size (Table 3.1).

![Figure 3.7](image)

**Figure 3.7** Aggregation index of forest cover in study areas where this index exhibited a decline (Veracruz, Salta and central Chile) or was constant (Chiapas).
Changes in the spatial patterns of the dryland forests are explained largely by changes in neighbouring land-cover. Interestingly, in three study areas forest fragments were primarily surrounded by croplands while in the other three areas, the fragments were adjacent to degraded forest (in the case of Salta and central Chile) or to shrubland (in Bariloche) (Figs. 3.13 and 3.14). In most of the study areas the dryland forest fragments were surrounded by more than 70% croplands or by degraded forest and shrubland. This high percentage of adjacency to human-induced land uses indicates that most forest edges may be subjected to anthropogenic activities that could potentially affect the survival of many species.

Oaxaca and Veracruz showed greater dynamics in the percentage of adjacency between croplands and forest fragments (Fig. 3.9) than in study areas surrounded by degraded forest or shrubland (Fig. 3.10). The greater variation in Oaxaca and Veracruz is related to changes in the matrix composition represented by a replacement of grassland and bare ground by cropland, particularly during the 1970s and 1980s. Later in the 1990s, the adjacency to cropland tended to decline in Veracruz owing to the expansion and replacement of cropland by grassland. In Chiapas and in Oaxaca, more than 90% of forest fragments were by cropland during the last decade (Fig. 3.9). On the other hand, in all the South American study areas, forest patches were largely surrounded by degraded forest or shrubland (Fig. 3.10). In Salta a gradual decline in the adjacency between forest and degraded forest was observed after the mid-1980s. In central Chile, which recorded the greatest adjacency to degraded forest, a kind of 'pseudo-savannah' named espinal was reached by 2000, which then declined by 2008 (Fig. 3.10). This variation was related to the dynamics of the espinal that was converted to cropland, and which originated from the degradation of sclerophyllous forest.

Figure 3.8  Aggregation index of forest cover in study areas where this index exhibited an increase (Bariloche and Oaxaca).
Figure 3.9  Percentage of adjacency between dryland forest patches and cropland in Chiapas, Oaxaca and Veracruz. Cropland was the major land-cover type adjacent to forest patches in these study areas.

Figure 3.10  Percentage of adjacency between dryland forest patches and degraded forest/shrubland in Salta, Bariloche and central Chile. Degraded forest and shrubland were the major land-cover types adjacent to forest patches in these study areas.
Landscape states and habitat destruction

McIntyre and Hobbs (1999) discuss the process of habitat destruction and habitat modification, which can be conceptualized as a continuum, associated with the influence of human disturbance. They identified four types of landscape states along a gradient of destruction: intact (<10% habitat destroyed), variegated (10–40%), fragmented (40–90%) and relictual (>90%). In variegated landscapes, the habitat still forms the matrix, whereas in fragmented landscapes, the matrix is composed of destroyed habitat. During the study period, the study areas exhibited different degrees of forest loss (Fig. 3.3 and Table 3.2). In Veracruz, the percentage of remaining forest cover (see Table 2.1 in Chapter 2 on recent loss) was 9% in 2000, which corresponds to a relictual landscape. In this area the effect of intensive agricultural development has led to a progressive fragmentation of forest habitat and to a high rate of forest loss (Fig. 3.5).

On the other hand, the percentages of forest cover detected were 16% in Bariloche, 32% in Chiapas, 35% in central Chile and 60% in Oaxaca. These study areas correspond to fragmented landscapes characterized by more than 40% of habitat destroyed. In Oaxaca and central Chile, most of the forest fragments are under high pressure from an intensively used matrix, which has led to a progressive transformation towards agriculture and degraded forest respectively (Fig. 3.10 and Table 3.2). Central Chile suffered the largest reduction of forest habitat from 43% to 34% of forest cover and has been severely fragmented (Fig. 3.12). In contrast, in Bariloche and Chiapas the native forest fragments remain relatively unmodified and the forest persisted and even defragmented after long periods of grazing by livestock, expansion of crops and fires. Salta was found to be in an intact state in 1977 (94% forest cover) and then changed to a variegated state in 2006 (73%). In this study area, the dryland forest still forms the landscape matrix and is represented by large patches (Fig. 3.9). However, forestry operations in these forests impose slight but continuous changes in the spatial patterns of forest cover, reflected by a reduction in the degree of aggregation (Fig. 3.7). The identification of the level of modification is an important consideration for management planning, as this can assist in deciding where and when to allocate greater and lesser protection to the landscape (Hobbs, 2002).

Trends in spatial patterns in the dryland forest in Latin America

The analysis of the landscape indices enabled the spatial patterns of dryland forests in Latin America to be assessed. Our results show that the spatial patterns of change of dryland forest were dynamic and did not necessarily represent a unidirectional process of forest loss and fragmentation (Table 3.2). This is consistent with what has been described for some other landscapes, emphasizing that there is no single correct way in which to think about spatial patterns in modified landscapes (Lindenmayer and Fischer, 2006). By recognizing the diversity of landscape trends, conservation strategies could be better focused (McIntyre and Hobbs, 1999).

In central Chile, the spatial patterns are related to a unidirectional landscape alteration, with continued alteration assumed to reduce both the size of individual patches (known as shrinkage) and the overall number of patches (known as attrition) (Forman, 1995b) (Table 3.1). Veracruz and Salta also showed a unidirectional alteration, but with an opposite trend in the number of patches that increased over time, defined as fragmentation (Forman, 1995a) (Table 3.2). On the other hand, Oaxaca showed a bidirectional alteration characterized by a rapid
fragmentation in the earliest years followed by a loss of forest patches in the last time interval (Table 3.2). This trend was more evident owing to a rapid increase in the number of patches during the first time interval (Table 3.1). However, there are many cases where trends in landscape change have been reversed (Metcalfe and Bradford, 2008; Vellend, 2003; Wittenberg et al., 2007). Bariloche showed an increase in patch size and increase in patch proximity over time (Table 3.3). In this area, the spatial patterns changed during the study period because of forest regeneration in logged areas. Chiapas showed a more stable pattern of change, increasing slightly the size of forest fragments and reducing the number of patches (Table 3.1).

Our results demonstrate that the spatial patterns of dryland forests were highly dynamic over the last four decades. While most of the study areas experienced a reduction of forest habitats others showed an increase or stability in forest cover. Understanding the trends in spatial patterns of dryland forest is important for the conservation of its biodiversity and the provision of diverse ecosystem services. Despite this importance, many forest assessments and international initiatives still focus on the extent of forest loss without concern for its spatial pattern (Kupfer, 2006). This work confirms further the advantages of using landscape metrics to describe pattern change, as has been demonstrated in other parts of the world (Bhattarai et al., 2009; Cayuela et al., 2006; Martínez et al., 2009; Peng et al., 2010; Trani and Giles, 1999; Zeng and Wu, 2005).

Table 3.2  Landscape states and trends in the spatial patterns of dryland forests in six study areas in Latin America over the last four decades.

<table>
<thead>
<tr>
<th>Study area</th>
<th>Description</th>
<th>Trend in spatial patterns</th>
<th>Landscape state</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salta</td>
<td>Division of large forest patches, increasing number of patches, decrease in forest aggregation</td>
<td>Progressive fragmentation</td>
<td>Intact to variegated</td>
</tr>
<tr>
<td>Veracruz</td>
<td>Loss of forest cover; decrease and increase in forest aggregation; substantial changes in matrix composition</td>
<td>Fragmentation followed by deforestation</td>
<td>Fragmented</td>
</tr>
<tr>
<td>Oaxaca</td>
<td>Loss of forest patches; and forest continuity</td>
<td>Progressive deforestation</td>
<td>Fragmented</td>
</tr>
<tr>
<td>Chiapas</td>
<td>No spatial changes in forest cover</td>
<td>Forest persistence</td>
<td>Fragmented</td>
</tr>
<tr>
<td>Bariloche</td>
<td>Union of forest patches; increase in forest aggregation</td>
<td>Forest patch coalescence</td>
<td>Fragmented</td>
</tr>
</tbody>
</table>

**Mapping forest degradation**

Human disturbances not only alter the spatial patterns of forest cover but can also lead to a modification or degradation of the remaining habitat (McIntyre and Hobbs, 1999). Modifications include changes to the structure, biotic composition, or ecosystem functioning of habitat (McIntyre and Hobbs, 1999; Ravi et al., 2010). Degradation of dryland forest habitat is associated with diverse human activities such as livestock grazing, tree harvesting and changed fire regimes (Reynolds et al., 2007).
MODIS data were used to map forest degradation in central Chile between 2002 and 2009 (Box 3.3). Owing to the low spatial resolution of MODIS images (250 m), a threshold of 3m²/m² in Leaf Area Index (LAI) was used to select only dense forests. This enabled other land-cover types such as pasture or shrubland to be excluded in order to monitor the degradation of forest cover. Then, changes in red and infrared band values extracted from MOD 13 Q1 between 2002 and 2009 were analyzed to detect pixels with a degree of degradation. Essentially, it was assumed that dense forest pixels with an increase in reflectance values for the red band and a decline in the infrared band over time correspond to pixels affected by degradation. Pixels without changes in reflectance values for both bands have not been affected by degradation. Degraded forest pixels were selected to determine the degree of degradation. This was conducted applying the NDVI (Normalized Difference Vegetation Index) for which three levels of degradation were defined: NDVI>0.71: low degradation; 0.57<NDVI<0.71: intermediate degradation; 0.57<NDVI: high degradation. These levels were validated in the field. Finally, an assessment of forests degraded during each time interval (2002–2005 and 2005–2009) was obtained by overlapping the corresponding binary maps of degraded forest/non-degraded forest cover in a Geographic Information System (GIS).

A logistic regression modelling approach was used to determine the immediate drivers of degradation. Most of the environmental and socio-economic explanatory variables that are potentially related to forest degradation were identified and mapped. These are: property size, slope, elevation, distance to roads, distance to rivers, distance to towns and distance to agricultural land in the earliest image.

In Chile, the forests degraded between each time interval (2002–2005 and 2005–2009) were identified by overlapping the corresponding binary maps of degraded forest/non-degraded forest cover in a GIS. In Argentina, degradation analysis was conducted for eroded premontane forest for each of the following time intervals: 1977–1987, 1987–1983, 1983–2006 and 1977–2006. The binary response variable, degraded habitat vs. non-degraded habitat, was analyzed using a logistic regression model in the statistical package R (Echeverria et al., 2008). In Salta, Argentina, degraded forest was derived from the Landsat image classification. This corresponded to forest areas with less than 50% of tree cover classified as eroded premontane forest with limiting edaphic factors. This included two types of savannah: (a) with edaphic restrictions and fire controlled, dominated by tusca blanca (Acacia albicorticata) and urundel (Astronium urundeuva), (b) tusca blanca and pasto cubano savannah, small patches of secondary forest (previously used for agriculture) and low stature riparian forests. The following were also included: eroded premontane forest, secondary forest and riparian forest. Degradation analysis was conducted for eroded premontane forest for each of the following time intervals: 1977–1987, 1987–1983, 1983–2006 and 1977–2006.

A total of 27,831 ha, equivalent to 28% of the total dense dryland forest in 2002, was degraded by 2009 in the Chilean study area (Fig. 3.11). Regarding the degree of degradation, the results demonstrated that the proportion of highly degraded forest increased in the second time interval. In 2005, 64% of the degraded forests were categorized as highly degraded, 23% as a moderately degraded and 14% as slightly degraded (Figs. 3.11 and 3.12). In 2009, 74% of the degraded forest was highly degraded, 24% moderately degraded and 2% slightly degraded (Figs. 3.11 and 3.12). This increase in the area of highly degraded forest can be explained by the fact that local people continued logging the dense forest for firewood and other forest products. These processes have caused a modification of the structure and composition of the remaining forests.
In Salta, 4.8% of the forest cover was degraded in 1977 (Fig. 3.13). This value remained almost constant during the following years, reaching 4.6% in 2006 (Fig. 3.14). A higher proportion of degraded forest occurred in 1987 with 5.3% of the forest cover. Between 1977 and 1987, forest degradation was mainly associated with livestock grazing, and forest logging for firewood and timber. At the beginning of the 1990s the rapid conversion of degraded forest to soya crops led to a decline in the area of degraded forest. During the last decade, the proportion of degraded forest remained almost constant, which reflects the current presence of degrading activities such as forest logging and browsing by livestock.

Figure 3.11 Distribution of degraded and non-degraded forest between time intervals in Chile. Degraded forest is shown at three levels of degradation.

Figure 3.12 Distribution of degraded forest by level of degradation between time intervals in Chile.
Figure 3.13  Maps of forest degradation in Salta, Argentina between 1977 and 2006.
Box 3.3  Estimating forest degradation in dryland landscapes in central Chile using MODIS products

D. González, R. Fuentes, C. Echeverría.

Forest degradation may occur owing to natural (forest fire, earthquakes, volcanism, etc.) and anthropogenic perturbations (urban and agriculture surface expansion, forest use, etc.) (Stuart et al., 2002; Pickett and White, 1985; Hüttl and Schneider, 1998). Nutritional disturbances can also lead to decreasing stand stability and productivity (Stolpe et al., 2008). Although dry forests are being subjected to a range of different disturbances, a major factor responsible for their loss and degradation is the recent expansion of industrial agriculture, resulting from increasing global food demand (Grau et al., 2009).

Remote sensing imagery becomes a powerful tool to evaluate the threats to forest ecosystems (Luque, 2000; Armenteras et al., 2003; Echeverría et al., 2007). The research described here focused on quantifying the degradation of a dryland forest in an area of 1,250,000 ha using MODIS satellite products. The study area is located in one of the most populated regions of dryland forest in Chile, between 33° and 38° S latitude, located between the Central Valley and Coastal Range (Fig. 1).

Using the MOD15A2 product at 1000 m spatial resolution, for three years (2002 (t0), 2005 (t1) and 2009 (t2)), we selected those forest patches whose pixels were equal to or greater than 3 m²/m² of leaf area index (LAI). This threshold enabled dense forest patches that may exhibit degradation to be distinguished. Those patches in which disturbances may have caused a removal of the forest cover were discarded as they represented deforestation instead of degradation.

Further, we applied the Near Infrared (NIR) and Red (R) reflectance responses from MOD13Q1 products at the pixel level to quantify the degradation over time. In a given forest pixel, when the NIR reflectance increases and the R reflectance decreases between two measurements (at the same daily time), the forest is increasing its canopy cover and, therefore, is becoming more dense. On the other hand, when the NIR reflectance goes down and R reflectance goes up, the forest is decreasing its canopy cover, which indicates that the forest patch is being degraded through time (Chuvieco, 1996). These responses in reflectances were modelled in ARC GIS applying a decision tree procedure (Fig. 2).

To visualize the levels of forest degradation, we related the Red and NIR behaviour with Normalized Difference Vegetation Index (NDVI) (Eq. 1). This was conducted applying the NDVI where three levels of degradation were defined: NDVI>0.71: low degradation; 0.57<NDVI<0.71: intermediate degradation; 0.57<NDVI: high degradation. All these levels were validated in the field.
Box 3.3 (cont.)

Eq. 1. NDVI=(δNIR-δRed) / (δNIR+δRed), where:

δNIR: Near Infrared reflectance

δRed: Red reflectance

Figure 1  Location of dense dryland forest in the study area in central Chile.

Figure 2  Decision tree for the quantification of forest degradation at the pixel-level.

A logistic regression modelling approach was applied to determine the immediate drivers of degradation. Most of the environmental and socioeconomic explanatory variables that are potentially related to forest degradation were identified and mapped. These include: property size, distance to roads, distance to rivers, distance to towns, distance to agricultural land in the earliest image and slope and elevation. Our results revealed that in 2005, 64% of the degraded forest surface was categorized as highly degraded, 23% as a moderately degraded and 14% as slightly degraded. In 2009, 74% of the degraded forest was highly degraded, 24% moderately degraded and 2% slightly degraded.
Box 3.3 (cont.)

Table 1  Percentage of degraded dense forest at the municipality level. Municipalities of at least 500 ha are listed here.

<table>
<thead>
<tr>
<th>Municipality</th>
<th>Degradation (%)</th>
<th>Municipality</th>
<th>Degradation (%)</th>
<th>Municipality</th>
<th>Degradation (%)</th>
<th>Municipality</th>
<th>Degradation (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Olmué</td>
<td>11.8</td>
<td>Navidad</td>
<td>25.6</td>
<td>Doñihue</td>
<td>28.7</td>
<td>Graneros</td>
<td>33.7</td>
</tr>
<tr>
<td>Limache</td>
<td>15.3</td>
<td>Coinco</td>
<td>26.7</td>
<td>San Antonio</td>
<td>29.8</td>
<td>Melipilla</td>
<td>34.6</td>
</tr>
<tr>
<td>Quilpue</td>
<td>19.4</td>
<td>Alhué</td>
<td>26.8</td>
<td>Buin</td>
<td>30.1</td>
<td>San Pedro</td>
<td>37.9</td>
</tr>
<tr>
<td>Casablanca</td>
<td>20.6</td>
<td>Valparaíso</td>
<td>27.8</td>
<td>Talagante</td>
<td>33.0</td>
<td>María Pinto</td>
<td>38.1</td>
</tr>
<tr>
<td>Santo Domingo</td>
<td>25.2</td>
<td>Paine</td>
<td>28.1</td>
<td>Curacavi</td>
<td>33.2</td>
<td>El Monte</td>
<td>42.5</td>
</tr>
<tr>
<td>Litueche</td>
<td>24.4</td>
<td>Rancagua</td>
<td>28.2</td>
<td>Las Cabras</td>
<td>33.5</td>
<td>Cartagena</td>
<td>46.9</td>
</tr>
</tbody>
</table>

The municipality with highest forest degradation percentage was Cartagena with 47% (Table 1). The municipalities that showed the lowest forest degradation percentages were Olmué (12 %), and Limache (15%) (Table 1). The multivariate logistic regression model, used to identify the main drivers of the landscape change process, indicated that the probability of an area being degraded is highly significant and positively related to the distance to streams. In contrast, the distance to urban areas and the distance to agricultural land in 2008 were negatively related to forest degradation. These results showed that the probability of degradation increases in forests located near urban areas, near agricultural land and far away from streams. In these areas, forest logging for fuelwood and browsing by cattle in dense forest are more intense causing a decline in canopy cover and tree density. Human access to dense forest appeared to be one of the main drivers of forest degradation.

The temporal analysis of the forest canopy changes based on NIR and red reflectance behaviour appears to be a suitable procedure to evaluate the degradation of dense dry forests. The main limitation of this procedure relates to the spatial resolution of the MODIS products and the size of dense forest areas.
Causes of forest degradation

In central Chile, the multivariate logistic regression models indicated that the probability of an area being degraded is highly significant and positively correlated with distance to streams \( (p<0.001; \text{Table 3.2}) \). In contrast, distance to urban areas and distance to agricultural land in 2008 were negatively related to forest degradation \( (p<0.01; \text{Table 3.2}) \). These results showed that the probability of degradation increases in forests located near urban areas, agricultural land and far away from streams. In these areas, forest logging for fuelwood and browsing by cattle in dense forest are more intense, causing a decline in canopy cover and tree density. Human access to dense forest appeared to be one of the main drivers of forest degradation.

Similarly, the probability of degradation in Salta was positively related to distance to urban areas in all the time intervals and for the whole study period \( (p<0.001; \text{Table 3.2}) \). Elevation also was positively related to forest degradation in all of the study periods. Distance to streams was marginally significant in the first and the third time interval as well as during the overall study period. This is because most of streams are located in areas that are less accessible for people. The probability of forest degradation was positively explained by distance to villages in the first time interval as well as during the whole study period (1977–2006). Before the 1990s, the distance to secondary roads was significantly related to forest degradation. Since the 1990s this variable has not been significant as most of the degraded forests near secondary roads were converted to agricultural land. During the 1990s, the distance to agricultural land was associated with the presence of degraded forests owing to the use of fire to expand the agricultural frontier.

In both study areas, the results revealed that accessibility to forest areas is one of the main drivers of forest degradation. The probability of a forest area being degraded is higher when a forest is located near urban and agricultural lands, and in lowlands. This trend reflects that the remaining dryland forests in central Chile and in Salta have been undergoing continuous degradation over recent decades (Box 3.4). This is consistent with the results obtained by recent assessments (MEA, 2005; Ravi et al. 2010), which demonstrate that dryland ecosystems around the world are undergoing rapid land degradation as a result of anthropogenic disturbances. Diverse studies emphasize that modifications of dryland forest habitats may lead to changes in ecosystem processes (Jafari et al., 2008; Smet and Ward, 2006; Stolpe et al., 2008), which may affect the productivity of the landscape, with important environmental and socioeconomic implications.
Box 3.4 Human-caused forest fires in Mediterranean ecosystems of Chile: modelling landscape spatial patterns of forest fire occurrence

A. Altamirano, C. Salas, V. Yaitul, A. Miranda, C. Smith-Ramírez

Fire disturbance is recognized as an important problem because it can devastate natural resources and human property, and threaten human lives. Forest fires result in enormous economic losses because they affect environmental, recreational, and amenity values as well as consume timber, degrade real estate, and generate high costs of suppression. Modelling forest fire occurrence (i.e., where and when a forest fire starts) has recently been conducted in the northern hemisphere (Calef et al., 2008; Lozano et al., 2007; Ryu et al., 2007; Vega-Garcia and Chuvieco, 2006), however, efforts on the subject are lacking for the southern hemisphere, in particular for Chilean ecosystems. Some studies in Chile have focused on post-fire effects on vegetation dynamics (Navarro et al., 2008; Litton and Santelices, 2003), but studies on predicting forest fires occurrence are lacking. In Chile it has been reported that fire can encourage exotic plant invasions (see Chapter 8) and cause significant losses of local biodiversity. Forest fire occurrence has increased in recent years in Chile, with a mean frequency of about five thousand forest fires per year. These fires have affected a mean area of about 500 km² per year (Navarro et al., 2008; CONAF, 2009), human activity being the main cause of fire ignition (CONAF, 2009). The extensive fires produced by human activity in central Chile need to be addressed in order for forest restoration approaches to be effective.

At the landscape scale (i.e. extents >100,000 ha), the probability of a large fire is associated with multiple factors including: forest type, physiographic characteristics, climate, and human activities. In this study we developed models to investigate the relationship between forest fire occurrence and landscape heterogeneity spatial patterns in Mediterranean ecosystems of Chile. The study area extends over 892 km² and is located in eastern central Chile covering parts of the Valparaíso and Metropolitan administrative regions (Fig. 1). We selected a landscape with temporal stability in composition operating at the landscape scale. We used georeferenced forest fire data from a 5-year period of fire occurrence from 2004 to 2008. A distance of 25 x 25 pixels (750 m) was used to compute the co-occurrence matrices, since small windows result in very sparse matrices. Our data on landscape spatial patterns were obtained at multiple spatial scales, including climatic, topographic, human-related, and land-cover variables from satellite imagery. We fit a logistic model in order to predict forest fire occurrence as a function of our potential predictor variables. The relationship modeled was that between the binary response variable (one = burned, zero = not burned) and the predictor variables. In order analyze forest fire occurrence we produced categorized maps of the predicted forest fire occurrence probability into four levels: very high (0.75 ≤ p ≤ 1), high (0.5 ≤ p ≤ 0.75), low (0.25 ≤ p ≤ 0.5) and low (0 ≤ p ≤ 0.25).

Our best model suggests that the probability of forest fire occurrence is related to both high temperature and precipitation, and lower distance to cities. Our predictions suggest that 46% (410 km²) of the study area has high probability of forest fire occurrence, being concentrated in the eastern locations of the study area (Fig. 1). Our model correctly classified about 73% of our validation dataset. The information from this study may be useful for hazard reduction, indicating risk of forest fire occurrence (Ryu et al., 2007; Vega-Garcia and Chuvieco, 2006). The study area is one of the most populated regions of Chile. Therefore, our findings can be used to inform decision making regarding land and urban planning. If climate determines patterns of forest fire occurrence, then when the climatic variables change, forest fire occurrence may also change. This might have important consequences for long-term land and urban planning, since prioritization of areas with high probability of forest fire occurrence today might not be effective in the face of climate change. Exploring a new statistical model approach would allow to improve the predictive capability of the models. Therefore, part of our future research will focus on this subject.
Conclusions

Our results indicate that dryland forests exhibit a progressive fragmentation and degradation in most of the Latin American landscapes studied during the research. In central Chile and in Salta, dryland forests have been simultaneously affected by forest loss (Chapter 2), fragmentation and degradation. In Veracruz and Oaxaca, the landscape has experienced a continuous fragmentation and loss of forest habitats. On the other hand, Chiapas and Bariloche show different trends, towards forest persistence and coalescence respectively. Results presented here clearly show that dryland forest is under considerable human pressure from economic development and imply policy challenges for the countries involved. Owing to the importance of dryland forest for providing different ecosystem services for human well-being, diverse actions should be undertaken to minimize or reverse the human impacts of fragmentation and degradation on dryland forests. Ecological restoration actions have the potential to address both the fragmentation and degradation of forest that has been documented here in multiple study areas. Such interventions should be planned and implemented at the landscape scale, to ensure they are effective in increasing connectivity among forest patches. Recent advances emphasize the development of integrative approaches to counter land degradation, poverty, safeguard biodiversity and protect the culture of the 2.5 billion people who live in dryland systems (Reynolds et al., 2007). Forest landscape restoration actions should constitute an element of such approaches. Urgent and comprehensive reframing of rural development strategies in Latin America should be undertaken to achieve this goal.
References


Assessing fragmentation and degradation of dryland forest ecosystems


Assessing fragmentation and degradation of dryland forest ecosystems


