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## Rapid deforestation and fragmentation of Chilean Temperate Forests

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### ABSTRACT

The temperate forests of Chile are classified a biological “hotspot” as a result of their high species diversity and high endemism. However, they are being rapidly destroyed, with significant negative impacts on biodiversity. Three land-cover maps were derived from satellite imagery acquired over 25 years (1975, 1990 and 2000), and were used to assess the patterns of deforestation and forest fragmentation in the coastal range of south-central Chile. Between 1975 and 2000, there was a reduction in natural forest area of 67% in the study area, which is equivalent to an annual forest loss rate of 4.5% per year using a compound-interest-rate formula. Forest fragmentation was associated with a decrease in forest patch size, which was associated with a rapid increase in the density of small patches (<100 ha), and a decrease in area of interior forest and in connectivity among patches. Since the 1970s, native forest loss was largely caused by an expansion of commercial plantations, which was associated with substantial changes in the spatial configuration of the native forests. By 2000, most native forest fragments were surrounded by highly connected exotic-species plantations. The assessment of forest loss and fragmentation provides a basis for future research on the impacts of forest fragmentation on the different component of biodiversity. Conservation strategies and land use planning of the study area should consider the spatial configuration pattern of native forest fragments and how this pattern changes over time and space at landscape level.

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## 1. Introduction

Habitat fragmentation and forest loss have been recognized as a major threat to ecosystems worldwide (Armenteras et al., 2003; Dale and Pearson, 1997; Iida and Nakashizuka,

1995; Noss, 2001). These two processes may have negative effects on biodiversity, by increasing isolation of habitats (Debinski and Holt, 2000), endangering species, and modifying species' population dynamics (Watson et al., 2004). Fragmentation may also have negative effects on species

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richness by reducing the probability of successful dispersal and establishment (Gigord et al., 1999) as well as by reducing the capacity of a patch of habitat to sustain a resident population (Iida and Nakashizuka, 1995). For example, fragmentation of Maulino temperate forest in central Chile has affected the abundance of bird richness (Vergara and Simonetti, 2004) and regeneration of shade-tolerant species (Bustamante and Castor, 1998), and has also favoured the invasion of alien species (Bustamante et al., 2003). The ecological consequences of fragmentation can differ depending on the pattern or spatial configuration imposed on a landscape and how this varies both temporally and spatially (Armenteras et al., 2003; Ite and Adams, 1998). Some studies have shown that the spatial configuration of the landscape and community structure may significantly affect species richness at different scales (Steiner and Köhler, 2003). Other authors emphasise the need to incorporate the spatial configuration and connectivity attributes at a landscape level in order to protect the ecological integrity of species assemblages (Herrmann et al., 2005; Piessens et al., 2005).

The temporal evaluation of forest change based on satellite imagery linked to fragmentation analysis is becoming a valuable set of techniques for assessing the degree of threat to ecosystems (Armenteras et al., 2003; Franklin, 2001; Imbernon and Branthomme, 2001; Luque, 2000; Sader et al., 2001). A number of deforestation studies have been conducted in tropical forests (Imbernon and Branthomme, 2001; Sader et al., 2001; Skole and Tucker, 1993; Steininger et al., 2001; Turner and Corlett, 1996) and, in particular, in the Amazon, which is now considered as the most studied region in the world by some researchers (Jorge and Garcia, 1997; Laurance, 1999; Laurance et al., 2000; Pedlowski et al., 1997; Ranta et al., 1998; Sierra, 2000). Conversely, few studies of deforestation and fragmentation have been reported for temperate forests (Staus et al., 2002), particularly in the southern hemisphere.

Chile has the largest temperate forest area in South America and more than half of the total area of temperate forests in the southern hemisphere (Donoso, 1993). Most of these forests are distributed along the Coastal and the Andean Range of Chile from 35° to 56° S and extend to a total of 13.4 million ha in the country (Conaf et al., 1999a). The temperate forest of Chile has been classified a biodiversity hotspot for conservation (Myers et al., 2000) and has also been included among the most threatened eco-regions in the world in the Global 200 initiative launched by WWF and the World Bank (Dinerstein et al., 1995). In these forests, a 34% of the plant genera are endemic (90% monotypic) (Armesto et al., 1997). However, Chile's temperate forests are being harvested to supply the increasing global demand for wood and paper products. A substantial amount of forest has also been lost due to the conversion of native forests to pasturelands, human-set fires, high grading (selective felling) and other logging practices (Lara et al., 2000).

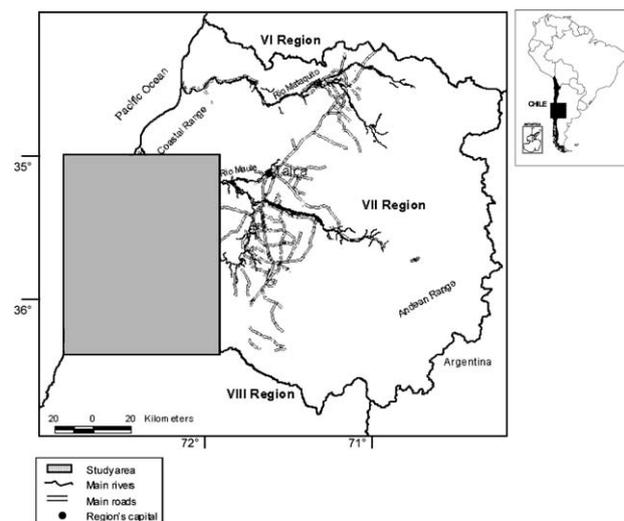
Although some of the ecological consequences of forest fragmentation have been studied in Chile (Bustamante and Grez, 1995; Donoso et al., 2003; Vergara and Simonetti, 2004; Willson et al., 1994), integrated spatial and temporal analyses have not been conducted. Although some attempts have been made in Chile to estimate the rate of deforestation (Lara et al., 1989; Olivares, 2000) or to assess land cover change (Conaf

et al., 1999b; Sandoval, 2001), these have been undertaken at local scales, over short time intervals (no more than 10 years), or using different types of data to compare forest cover over time, which confers some methodological limitations. Long-term analyses of the spatial patterns of deforestation and fragmentation of temperate forest ecosystems at the landscape scale have not yet been reported either in Chile or elsewhere in the southern hemisphere.

The purpose of this study is to contribute to the understanding of the patterns of deforestation and fragmentation in the temperate forests of Chile at the landscape level. In particular, we examined the patterns of land-cover change and the changes in the spatial configuration in the Maulino temperate forest over time and space by using satellite scenes acquired at different time intervals. In this study, we hypothesised that there has been a substantial loss of native forest as a result of an increase in area of exotic-species plantations. Also, we anticipated that this forest loss was related to fragmentation of Maulino forests due to changes in the spatial configuration in terms of size, shape and degree of isolation of forest patches. This work is the first step to understand the potential ecological effects of fragmentation and the proximate drivers and causes of deforestation, which will be addressed in other studies.

## 2. Study area

The Rio Maule-Cobquecura study area covers approximately 578,164 ha of land located in the Coastal Range of the Maule and Bio-Bio regions in south-central Chile (Fig. 1). The studied landscape partially includes the municipalities of Constitución, San Javier, Quirihue and Cobquecura, and all the area of the municipalities Empedrado, Chanco, Pelluhue, and Cauquenes. The area is characterised by rainfall concentrated during the winter that leads to dry summers from September to April with little cloud cover and high luminosity. The natural forest is mainly dominated by secondary forest of *Nothofagus* species (*N. obliqua* and *N. glauca*) (Fagaceae) and



**Fig. 1 – Location of study area in the coastal range, between Rio Maule and Cobquecura.**

sclerophyllous species including *Acacia caven* (Mimosaceae), *Quillaja saponaria* (Rosaceae), and *Maytenus boaria* (Celastraceae). Also, many endangered species such as *N. alexandri*, *Pitavia punctata* (Rutaceae), and *Gomortega keule* (Gomortegaceae) occur in the study area.

Forest clearance on a significant scale began with the arrival of European colonizers in the XVI–XVIIth century. Then, in middle of the XXth century a boom in the cultivation of wheat crops resulted in the elimination of extensive forest areas in the Coastal Range of the study area (San Martín and Donoso, 1997). In recent decades, the use of native forests for firewood has led to extreme degradation of this forest resource (Olivares, 2000). According to field observations and different surveys, most of the native forests in the study area correspond to highly degraded secondary forest (Donoso and Lara, 1995). This type of fragmented forest is severely impoverished in commercially valuable timber species as a result of selective logging and fire wood extraction. Some of the remnant forest fragments are areas that were clear-felled for shifting cultivation and subsequently abandoned (Lara et al., 1997).

### 3. Methods

#### 3.1. Remote sensing data

To analyse changes in forest area and spatial pattern over time, a set of three Landsat scenes were acquired for the years 1975 (MSS), 1990 (TM), and 2000 (ETM+). In order to carry out a quantitative comparison of the images in the present study, the original 79 m MSS raster grids were resampled to the resolution of the TM and ETM+ raster grids (30 m) (Staus et al., 2002; Steininger et al., 2001). This process was made using the re-project algorithm of PCI (2000). The fine grain used here (30 m) allowed the identification of non-forest areas within forest patches, which is an important spatial attribute to identify in areas affected by high-grading (selective felling). The presence of small patches is an important attribute to quantify in forest fragmentation analysis, but can only be assessed using high-resolution imagery (Millington et al., 2003). Although some of these small fragments may be important for the conservation of some species (Greze, 2005), a minimum mapping unit of greater than 5 pixels were used in this study. This enabled differences in data quality produced by the resampling of the MSS images to be minimised.

#### 3.2. Pre-processing of the satellite data

It was necessary to correct the images geometrically, atmospherically and topographically before they could be used to assess changes in forest cover and fragmentation (Chuvieco, 1996; Rey-Benayas and Pope, 1995). Geometric correction was performed using the “full processing” module in PCI Geomatics. This consisted of a transformation of each image using both GCPs (ground control points) and a 2nd order polynomial mathematical model. ETM+ images were spatially corrected in order to use them as a basis to correct the MSS and TM images. The satellite images were georeferenced separately to vector maps by locating approximately 55–65 corresponding GCPs in each image and the reference map. Road

networks, based on topographic maps digitalized in 1970s, were used to correct the 1976 and 1975 images. The geometric accuracy ranged from 0.10 to 0.39 pixels, corresponding to 3–11.7 m. Atmospheric correction was applied to all the scenes transforming the original radiance image to a reflectance image (Chávez, 1996). The topographic correction was performed for each scene using the method proposed by Teillet et al. (1982) in order to remove shadows in hilly areas.

#### 3.3. Image classification

Four resources were available to aid image classification. “Catastro” is a GIS-based data set of thematic maps derived from aerial photographs and satellite imagery between 1994 and 1997 (Conaf et al., 1999a). This data set provided detailed information on land use and forest types (including dominant tree species, forest structure, and degree of disturbances) at 1:50,000 scale considering a minimum mapping unit of 6.25 ha. Catastro was used both to define the land cover types for the present study and for the image classification of the 2000 ETM+ scene. A second set of data comprised 11 digital aerial photographs at 1:115,000 taken in 1999 (Conaf and Uach, 2000), which were also used to the image classification of the ETM+ image. A third data set corresponded to forest cover maps generated from aerial photograph at 1:60,000 between 1978 and 1987 (Lara et al., 1989). These maps were used for the image classification and for the accuracy assessment of the earliest images. A four reference group corresponded to 65 control points sampled in field visits between 2001 and 2002. Information on the history of land cover change for the points visited was also collected for the interpretation of the images, particularly for the earliest ones.

Owing to the availability of these ground-based data sets, a supervised classification was the method chosen to classify the three Landsat scenes. The statistical decision criterion of Maximum Likelihood was used in the supervised classification to assist in the classification of overlapping signatures, in which pixels were assigned to the class of highest probability. The selection of training sites was done considering representation of all digital categories of radiance according to the numeric values (spectral signature) and colour composites (Chuvieco, 1996). Some of these training areas were consistently delineated in each scene in order to minimise classification errors when performing change detection (Luque, 2000). Signature separability was assessed by the Bhattacharyya distance which is used to analyse the quality of training sites and class signatures before performing the classification.

Accuracy assessments of the MSS (1975) and TM (1990) images were conducted using aerial photograph-based land cover maps developed by Lara et al. (1989) for the years 1978 and 1987. Two new sets of 369 and 360 points were used for this purpose for each image respectively. The points were overlain on the reference land cover maps and assigned to the respective class. Confusion matrices were constructed to compare the class identified for each sample point with the land covers derived from the satellite images (Appendix 1). The accuracy of the ETM+ image was assessed by ground-truthing of 226 points visited between 2002 and 2003 (Appendix 1).

### 3.4. Land cover types

The following basic categories of land cover were identified from each image: (1) agricultural land (three sub-categories), (2) shrubland, (3) arboreal shrubland (an intermediate successional stage between shrubland and secondary forest with dominance of sclerophyllous species), (4) secondary forests (composed mainly of *Nothofagus* species such as *N. obliqua*, *N. glauca* and *N. alessandri*), (5) exotic-species plantation (mainly *Pinus radiata*), (6) young or new plantation, (7) wetland, (8) bare ground, (9) urban areas, and (10) water bodies. All forest cover in the study area was classified as secondary forests, owing to the absence of primary forest formations in the landscape examined.

### 3.5. Deforestation and fragmentation analysis

Maps were analysed using ARC VIEW (version 3.2; ESRI (1999)) and its extension Arc View Spatial Analyst 2.0 for Windows to quantify land cover change and forest loss and to configure grid covers for the application of landscape spatial indices. These indices (or spatial metrics) were computed by FRAG-STATS (version 3) (Mcgarigal et al., 2002) to compare the spatial pattern of forest cover for each time interval and study area. The native forest category was used to conduct the analysis of deforestation and fragmentation of native forests. In this study, the compound-interest-rate formula was used due to its explicit biological meaning (Puyravaud, 2003):

$$P = \frac{100}{t_2 - t_1} \ln \frac{A_2}{A_1}$$

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

Quantification and comparison of the spatial configuration of native forest fragments was conducted based on the following set of key landscape metrics selected after reviewing recent forest fragmentation studies (Armenteras et al., 2003; Franklin, 2001; Fitzsimmons, 2003; Imbernon and Brant-homme, 2001; Millington et al., 2003; Staus et al., 2002; Steininger et al., 2001): (a) patch area (ha), (b) patch density (number of patches per 100 ha), (c) largest patch index (% of the landscape comprised by the largest patch), (d) total edge length (km), (e) total core area (total patch size remaining after removing a specific buffer edge) (ha), (f) mean proximity index (ratio between the size and proximity of all patches whose edges are within a specified search radius of the focal patch [250 m, 1 km and 2 km]), (g) radius of gyration (average

extensiveness in meters of connected cells), (h) aggregation index (% of like adjacencies between cells of the same patch type), and (i) adjacency index (length in kilometres of edge between native forest and other cover types). For the calculation of core area, the interior native forest was defined at different edge distances: 100, 300 and 500 m. These distances enabled more properly the interior area to be assessed under different scenarios of edge effects. Most of these metrics have been applied in diverse types of landscapes and have enabled the assessment of spatial attributes in fragmented landscapes. A Kruskal-Wallis test was used to determine if the indices obtained between time intervals were different at the 95% confidence level (Dytham, 2003).

## 4. Results

### 4.1. Accuracy assessment

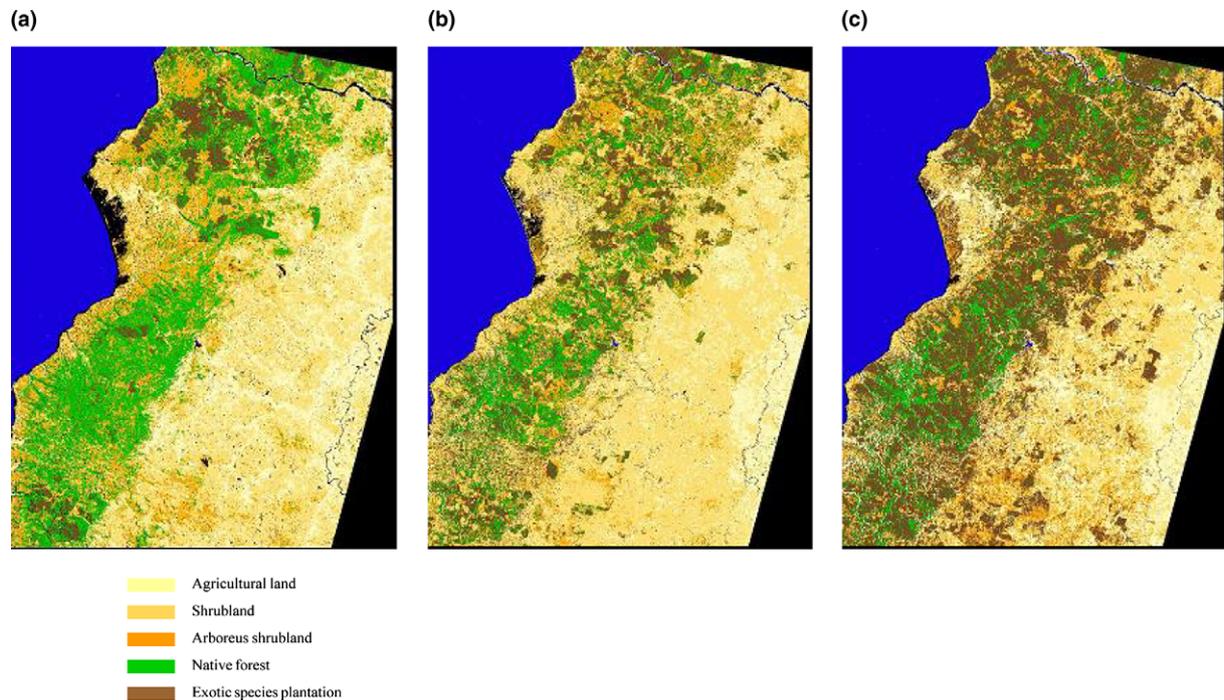
According to the rules suggested by PCI geomatics for ranges of separability values of class signatures, a “good” signature separability was achieved for each image classified, the average Bhattacharyya distance ranging from 1.979 to 1.987. Overall agreement of classification was 82.7% for the 1975 MSS, 83.3% for 1990 TM image, and 84.9% for the 2000 ETM+ image (Appendix 1). The lowest values of accuracy corresponded to shrubland, arboreal shrubland, and native forest categories. These three formations are very similar in their spectral signatures (Bhattacharyya distance less than 1.9) because they correspond to stages in a continuous succession process, which may easily produce misclassifications between the categories assigned in the training site and those classified by the algorithm. Supporting data from other studies and control points were important to distinguish these stages of forest succession.

### 4.2. Major changes in land cover

Changes in land cover were analysed using the area statistics (Table 1) derived from land cover type maps (Fig. 2). The estimated cover of native forests decreased from 119,994 ha in 1975 to 39,002 ha in 2000. In other words, 67% of the native forests existing in 1975 had been replaced by other land cover types by 2000. In 1975 the native forests were distributed throughout the study area. Twenty five years later, these formations were restricted to small patches, sparsely distributed

**Table 1** – Area of land cover types in 1975, 1990, and 2000 in Rio Maule-Cobquecura

Cover type	1975		1990		2000	
	(ha)	(%)	(ha)	(%)	(ha)	(%)
Agricultural and pasture lands	105,701	18	78,482	14	124,819	22
Shrubland	193,532	34	260,607	45	104,151	18
Arboreal shrubland	112,818	19	79,643	14	93,261	16
Native forest	119,994	21	56,133	10	39,002	7
Exotic-species plantation	29,579	5	96,777	16	211,686	36
Other land covers	16,541	3	6522	1	4800	1
Total	578,164	100.0	578,164	100.0	578,164	100.0



**Fig. 2 – Temporal and spatial variation of the major land cover types in Rio Maule-Cobquecura for the years: (a) 1975, (b) 1990, and (c) 2000.**

across the landscape (Fig. 2). Conversely, exotic-species plantations increased from 5% land area in 1975 to 17% in 1990; by 2000 this land cover type was the dominant cover on the map, comprising 36.6% of total land area. In 1975 exotic-species plantations were located principally in three specific areas within the study area. By 2000, the plantations had rapidly expanded across the landscape, reaching sites at different altitudes and aspects. Shrublands and arboreus shrublands were the dominating vegetation formations in the cover maps of 1975 (33%) and 1990 (58%). Ten years later, these formations decreased to represent 34% of total area, and the exotic-species plantation became the dominant land cover type at that time. During the first time interval, 36% of the native forest area was converted to shrubland and arboreus shrublands, 29% to exotic-species plantations, and 31% remained as native forest. During the second time interval, a substantial area (50%) of native forest was replaced by exotic-species plantations whereas only 7% was converted by logging to shrublands and arboreus shrublands and 36% remained as native forest. More than half (53%) of the native forest cover existing in 1975 has gradually been converted into exotic-species plantations by 2000; another substantial area (40%) was transformed into shrublands or arboreus shrublands. From 1975 to 2000, agricultural and pasture lands exhibited a slight increase (Table 1). However, in 1990 the area occupied by this cover type declined to 14% as a consequence of the conversion of land cover types (including pasture lands) to exotic-species industrial plantations after the promulgation of the law on forestation in 1976. Conversely, shrubland presented an increase from 1976 to 1990 due to the clearance of secondary forests. In 2000, most of this shrubland appeared to be covered by exotic-species plantations. By overlapping the forest native cover of each year, it was observed that between

36% and 44% of native forest was derived from regeneration of shrubland into secondary forest between time intervals respectively. However, it was noted that between 80% and 90% of this category corresponded to small patches whose area was less than 5 ha.

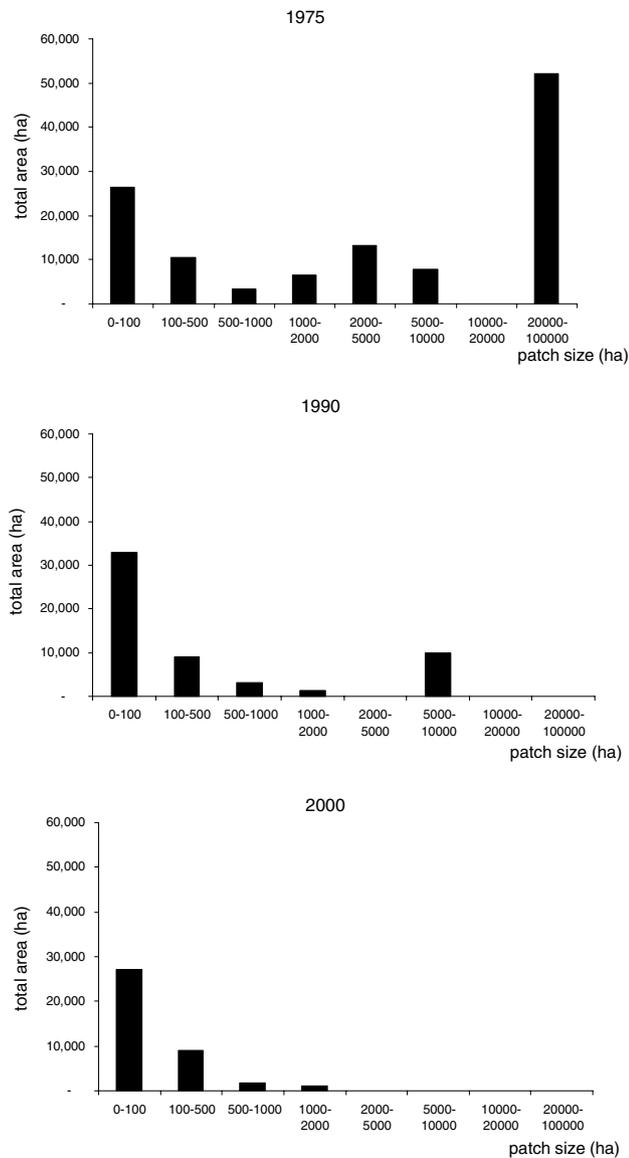
During the whole study period, the annual deforestation rate was 3240 ha year<sup>-1</sup>, equivalent to 4.5% year<sup>-1</sup> using the compound-interest-rate formula. Most of the forest loss was concentrated in the first 15 years of the study period, at a deforestation rate of 5.06% year<sup>-1</sup>, corresponding to 4257 ha year<sup>-1</sup>. Between 1990 and 2000, the rate decreased slightly to approximately 3.64% year<sup>-1</sup>, equivalent to 1713 ha year<sup>-1</sup>.

#### 4.3. Variation in forest patch size

One of the basic symptoms of forest fragmentation is the increase in number of smaller patches (Fig. 3). In the Rio Maule-Cobquecura considerable changes were found in the distribution of forest patch size between time intervals (Fig. 3). By 1975, 44% of the forest area was concentrated in a large patch between 20,000 and 100,000 ha; the remaining forest area occurred in isolated patches of less than 10,000 ha, with almost half occurring in very small patches of less than 100 ha. In 1990, 59% of the total area of native forests occurred in patches of less than 100 ha. By 2000, this percentage increased to 69% and only 3% had a size greater than 1000 ha.

#### 4.4. Spatial configuration of native forest cover

The highest density of patches was recorded in 1990 with 1.65 fragments of native forest per 100 ha, subsequently decreasing to 1.36 in 2000. During the first period, the native forests were mainly affected by severe fragmentation (increasing



**Fig. 3 – Temporal variation of forest fragment size in Rio Maule-Cobquecura.**

number of patches) and deforestation (decreasing patch size). By 2000 deforestation became the dominant process, owing to a decline in both mean patch size and patch density. The process of fragmentation was accompanied by the loss of the

largest forest patch (Table 2), ranging from 7% in 1975 to 0.2% of the total area in 2000. Continuous areas of quality habitats decreased following the introduction of disturbed fragments into the matrix. These modifications of the landscape were also characterised by the presence of more patch edges (Table 2). There was no significant difference in the total edge length ( $\chi^2_1 = 0.28$ ,  $P = 0.59$ ) between 1975 and 1990. Similarly, no significant difference was registered in the total edge length ( $\chi^2_1 = 2.31$ ,  $P = 0.12$ ) between 1990 and 2000. However, the absolute increase of edge length during the first time interval indicated that the shape of native forest patches became more irregular (Table 2).

No substantial difference between mean proximity indices calculated at different distances (250 m, 1 km, 2 km) was recorded. Following this result, a distance of 1 km was selected as this represents an intermediate value between the others. The native forest fragments registered a significantly different Mean Proximity Index between 1975 and 1990 ( $\chi^2_1 = 1353.8$ ,  $P < 0.001$ ) and between 1990 and 2000 ( $\chi^2_1 = 302.24$ ,  $P < 0.001$ ). The main change in the index was recorded at the first time interval. During this period, the neighbourhood of forest patches rapidly became occupied by areas of a different land cover type, as native forest patches became spatially separated and less contiguous in distribution. Similar temporal variation was observed for the radius of gyration in the study landscape (Table 2). Although this index is a measure of patch extent, it indirectly provides information about the structural connectivity of the landscape. This index varied significantly from 6630 m in 1975 to 1619 m in 1990 ( $\chi^2_1 = 351.9$ ,  $P < 0.001$ ). By 2000, this index had not decreased significantly ( $\chi^2_1 = 0.46$ ,  $P = 0.49$ ). Throughout the study period, the radius of gyration decreased by approximately 91%, which indicates a substantial decline in the extensiveness of connected cells of native forest fragments. In other words, this decreasing trend in the connectivity of forest fragments is the result of increasing forest fragmentation over time, which has dissected the native forests into more compact and smaller fragments.

The total amount of core area declined consistently over time at difference edge distances. For the edge buffer of 100 m, a sharp decline was observed from 21,138 ha in 1975 to 666 ha in 1990, and to 27 ha in 2000. For the 300 m and 500 m edge buffers the core areas fell respectively from 918 ha and 839 ha in 1975 to zero in 1990 and 2000. During the first time interval the largest native forest fragments were affected by a rapid division that was associated with an increase in edge length (Table 2).

**Table 2 – Changes in landscape pattern indices for the native forests in Rio Maule-Cobquecura in 1975, 1990, and 2000**

Landscape indices	1975	1990	2000
Patch density ( $n/100$ ha)	0.93	1.65	1.36
Largest patch index (%)	6.91	1.30	0.16
Total edge length (km)	20,330	22,337	15,799
Total core area (ha)			
100 m	21,138	918	839
300 m	666	0	0
500 m	27	0	0
Mean proximity	5880 ± 18,585	612 ± 3263	73 ± 294
Radius of Gyration (m)	6630 ± 220	1619 ± 107	542 ± 93

**4.5. Spatial relations between native forest cover and other land cover types**

In 1975 the major land cover types exhibited a high frequency of like adjacency between pixels of the same cover type (index of aggregation greater than 80%) (Fig. 4a). By 1990 shrubland was the cover type most aggregated in the landscape (86%), followed by agricultural land (70%), exotic-species plantation (63.4%) and native forest (62.7%). In 2000 the level of aggregation of shrubland decreased to 80%, while exotic-species plantation and agricultural land increased to 79% and 72%, respectively. Conversely, native forest decreased to 59%, the lowest level of aggregation in the landscape. In 1975 the index of aggregation revealed that the few large plantations were highly aggregated, supported by a visual inspection of the land cover map (Fig. 2). Then, by 1990 new areas of plantation were sparsely established in the landscape, decreasing the frequency of like adjacency. However, by 2000 the rapid expansion of plantations across the landscape led to a more aggregated and compact cover (Fig. 4a).

Native forest fragments were gradually becoming increasingly adjacent to different types of patches between 1975 and 1990. This is confirmed by the Adjacency index, which shows that in 1975 most of the native forests were mainly adjacent to shrublands, and less to exotic-species plantation and agricultural land. From 1990 to 2000 this situation changed substantially as a result of the expansion of exotic-species plantations, resulting in an increase in the adjacency between

this cover type and native forest fragments (Fig. 4b). During the first time interval, the native forests were mainly surrounded by well-aggregated, interspersed patches of shrublands. Then, by 2000, most of the edges of the native forest fragments were surrounded by well-aggregated patches of exotic-species plantations (Fig. 4b).

**5. Discussion**

The present study constitutes the most extensive analysis of deforestation and forest fragmentation ever conducted in Chile. The results demonstrate how changes in spatial patterns of the forested landscape may be assessed using multi-temporal satellite data, as has been achieved recently in other areas of the world (Tommervik et al., 2003; Rees et al., 2003).

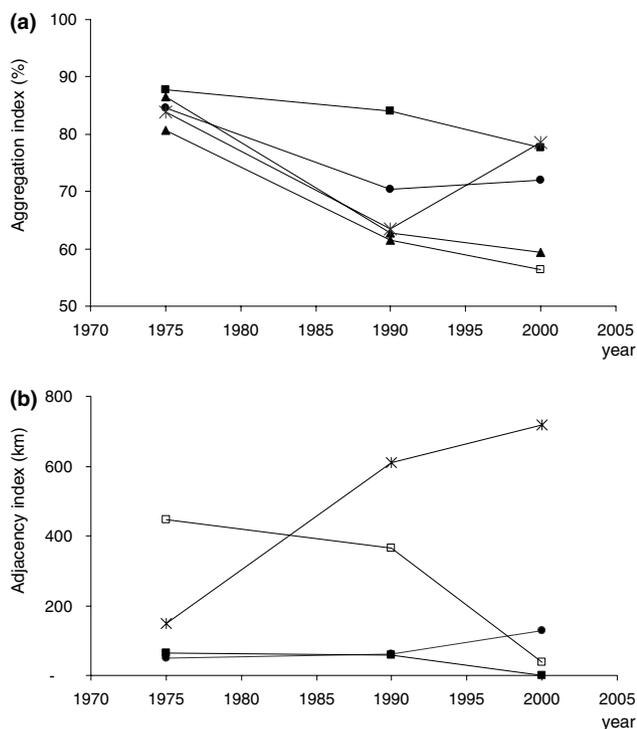
**5.1. Assessment of methods and results**

As classifications of satellite imagery into land cover types are never completely accurate, analyses of forest loss and landscape patterns of the present study are affected by errors in the classifications.

According to the confusion matrices (Appendix 1), image accuracy tended to improve as the image date became more recent. This trend may be related to image quality, which is better in TM and ETM+ images. For the original image, the accuracy assessment showed that the native forest cover is slightly overestimated which can affect the analyses derived from the 1975 map. However, this might be related to the availability of concurrent land cover maps for the earliest images. For instance, the accuracy of the 1975 map was assessed using aerial photograph-based maps for the year 1978. This difference in time may have caused a higher error commission than error omission, as an important area classified as native forest in 1975 had been converted to pine plantations or shrublands by 1978. This rapid conversion is more accentuated in the earliest images as this is when considerable land cover changes occurred in the study area.

The high percentages of overall accuracy of the images revealed that the supervised classification, which was strongly supported by ground-based information, provided a suitable identification of land cover types in each of the satellite scenes processed. Although the classification conducted for the oldest scene (MSS 1975) suffered from the disadvantage of limited ground validation, a cautious interpretation considering field control points that had not changed over time enabled high values of accuracy to be obtained, and clearly identifiable categories of land cover in the classification.

In order to manage these errors propagated through the analyses of change and patterns, some measures were adapted from Brown et al. (2000). In the present study, the error was minimised by applying improved topographic correction algorithms in the image processing and by aggregating some of the sub-categories of land cover types. Additionally, different ground-based surveys conducted from previous works and additional field visits were used to support the image classifications. Also, the error was minimised by filtering the classified image to remove small patches of less than four 30 m-pixels (equivalent to 0.36 ha).



**Fig. 4 – Temporal changes in (a) aggregation index and (b) adjacency index applied to the major land cover types in Rio Maule-Cobquecura. Land cover types: ● = agricultural land, □ = arboreous shrubland, ✱ = exotic-species plantation, ▲ = native forest, and ■ = shrubland.**

## 5.2. Forest cover change

The native forest of Rio Maule-Cobquecura has undergone high degrees of forest clearance during the three decades analysed here, compared to many other forested landscapes. Initially, the forest was severely deforested and degraded due to logging for fuelwood and clearance for cultivation (San Martín and Donoso, 1997). Forest losses recorded for other temperate forests, assuming that they were calculated using the compound-interest-rate formula, have generally been lower. A forest loss rate of 0.53% was estimated for the Klamath-Siskiyou ecoregion, USA, and an overall (cumulative) reduction of forest cover by 10.5% was recorded over the period 1972–1992 (Staus et al., 2002). In western Oregon, deforestation rates by clearcutting between 1972 and 1995 varied from 0.5% to 1.2% with almost 20% of the total forest impacted (Cohen et al., 2002). Similarly, in other areas of western Oregon, between 1972 and 1988 the rate of deforestation by primarily clearcutting was 1.2% of the entire study area including the wilderness area (Spies et al., 1994). In the central Sikhote-alin Mountains of the Russian Far East, an 18.3% total reduction in conifer forest and a 7.4% total reduction in hardwood forest cover were reported between 1973 and 1992 due mainly to disturbance by timber harvesting (Cushman and Wallin, 2000). A rate of 0.6%, much lower than that determined for the period 1990–2000 in the present study, was found for the years 1986 to 1996 in the Napo region of western Amazonia (Sierra, 2000). A higher rate of 6% was determined for lowland deciduous forest in eastern Santa Cruz, Bolivia in the middle 1990s (Steininger et al., 2001), thought to be one of the highest deforestation rates reported anywhere in the world.

Higher forest loss rate has been reported in an aerial photography-based study conducted for *Nothofagus alessandrii* forests, a threatened endemic tree to Chile, within the present study area (Bustamante and Castor, 1998). Using also the compound-interest-rate, this forest type declined at a rate of 8.15% between 1981 and 1991. If this rate remains constant, a total extinction of *N. alessandrii* forest is predicted by the authors in the next decade.

Lower rates of forest loss have been reported in other aerial photography-based studies conducted in the temperate forests of southern hemisphere, using the absolute-loss-rate formulae. For the pre-Andes of the Maule Region of Chile, Olivares (2000) reported deforestation rates in two study areas ranging from 0.5% to 1.4% between 1987 and 1996. For the Coastal Range of Chile, Lara et al. (1989) determined deforestation rates for the period 1978–1987 from 2.0% to 3.5% in the coast of the Maule (including Rio Maule-Cobquecura) and Bio-Bio regions respectively – closely comparable to the period 1990–2000 analysed for Rio Maule-Cobquecura (3.64%). Conversion to exotic-species plantations and clearance for agricultural land played a major role in these totals (Olivares, 2000; Lara et al., 1989). For the entire Maule region, a lower annual forest loss of 0.58% was determined between 1994 and 1999, using aerial photographs at difference mapping scales (Conaf and Uach, 2000). Further south, in northern provinces of the Lake region, an overall reduction of 18,100 ha, corresponding to an annual deforestation of 0.3% between 1995 and 1998 was obtained using remote sensing and aerial photographs (Conaf et al., 1999b). Different types of data and mapping scales have

been used at different measurement times in the studies mentioned above, which might have had an effect on the estimation of forest loss. The present study, in contrast, used a consistent type of data over a longer period.

An important increase in the area of shrublands was detected by 1990 as a result of the elimination of native forests and arboreal shrublands during 1975–1990. Agricultural and pastureland lands also expanded slightly by 2000 with the reduction of native forests and shrublands in flat areas situated under 200 m elevation.

## 5.3. Spatial pattern analyses

Landscape pattern indices provide a useful tool to explore in-site variability, cross-site differences and changes over time. The simultaneous use of class-level and patch-level landscape pattern indices enabled assessment of the spatial configuration of forest cover and its relation to principal land cover types. Conversely, Cumming and Vernier (2002) related only indices at class level to non-spatial metrics to simulate regional effects of forest management. Reed et al. (1996), in contrast, used only indices at landscape level to analyse forest fragmentation in south-eastern Wyoming, USA between 1950 and 1993.

Patch density and total edge length varied unpredictably with deforestation in Rio Maule-Cobquecura. These metrics increased in the earliest stage of forest loss and fragmentation and decreased during the later stages of deforestation. Zipperer et al. (1990) also observed that the constant action of deforestation led to a decline in patch density in central New York, USA. In the study area, this process even eliminated forest patches created in the first study period. Similarly to Ranta et al. (1998), the substantial increase of patch density in Rio Maule-Cobquecura was related to the concentration of the forest area in patches less than 100 ha in area. In our study, patch size declined consistently over time, which differs substantially to the situation recorded in Wisconsin (Pan et al., 1999). Armenteras et al. (2003) stated that progressive reduction in the size of forest habitats is a key component of ecosystem fragmentation.

The greatest absolute decline in the largest forest patch size coincided with the time period where the greatest absolute decline in annual forest loss was observed, as registered elsewhere (Fitzsimmons, 2003). This decline of large forest fragments might have a significant effect on the response of some species in the study area. For instance, the largest cloud forest fragments were the most important characteristic influencing the response of bird species in eastern Mexico (Martínez-Morales, 2005). Similarly, higher bird species richness of resident and migrant species occurred in larger forest fragments in Singapore Island (Castelletta et al., 2005).

Edge length exhibited different trends over time. For a similar period analysed, Fuller (2001) found that many of forest of the watersheds studied in Virginia, USA also exhibited most significant changes during 1987–1999. The edge length increased during the first time period of Rio Maule-Cobquecura, indicating that forest cover on average became more geometrically complex over time. Similarly, Fitzsimmons (2003) found for the perimeter-to-area ratio, a related index to shape index, inconsistent behaviour with deforestation between temporal and spatial comparisons. He suggested that this in-

dex may not be useful for establishing landscape management objectives.

The distance between patches is one the main factors affecting the colonization possibilities of species and, consequently, gene exchange between populations (Tischendorf and Fahrig, 2000). Some researchers have stated that the disruption of landscape connectivity – the functional linkages among habitat patches – may have substantial consequences for the distribution and persistence of population, and therefore with the maintenance of species diversity (Franklin, 1993; Rouget et al., 2003; With and Crist, 1995; Zebisch et al., 2003). The landscape studied here was affected over the last 25 years by a progressive separation of the forest patches due to a significant reduction of forest cover by conversion to other cover types. The mean proximity index experienced an abrupt decline between 1975 and 1990 owing to the division of large forest patches in the earliest stage of deforestation. This effect of deforestation was also observed by Imbernon and Brant-homme (2001) who selected the proximity index to characterise the degree of isolation and fragmentation in different tropical forest landscapes. The degree of isolation of 185 *N. alessandrii* fragments was analysed by estimating the distance between them within Rio Maule-Cobquecura (Bustamante and Castor, 1998). Although the temporal patterns of the isolation of this forest was not assessed in that study, results also revealed a certain isolation of the native forests which will prevent the genetic exchange of wind-dispersed plants.

As an overall measure of habitat connectivity (Keitt et al., 1997), the radius of gyration (or correlation length as known at the landscape level) indicated that landscape connectivity has significantly decreased over time in the study area. Trani and Giles (1999) reported that the connectivity index was not sensitive between contiguous and fragmented landscape conditions. However, they found that the contiguity index (an assessment of the unbroken adjacency of a forested landscape) declined with each stage of deforestation. In our study, the radius of gyration has enabled the connectivity of the native forest fragments at different time intervals to be quantified, taking into account the size and shape of the patches in the estimation of the index. This index clearly responded to a progressive deforestation, showing significant differences between time intervals. After its development in 1997 (Keitt et al., 1997), this index has not been apparently used to study the effects of forest fragmentation or deforestation. On the basis of the current results, the radius of gyration is highly recommended for analysis of the physical continuity of forest patches across the landscape and over time.

For the three different scenarios of edge distances, the Maulino forests of the study area exhibited a decreasing trend in the core area over time. For the Klamath-Siskiyou ecoregion (USA) the mean core area also declines substantially on public and private lands (Staus et al., 2002). This tendency was also observed in Chapare, Bolivia between 1986 and 2000, in which interior areas were defined by a distance of 300 m from the edge (Millington et al., 2003). The three scenarios of edge effect used in the present study can be interpreted as the magnitude of impacts of different ecological processes such as plant regeneration, seed dispersion, or plant growth and survival, due to forest fragmentation. In particular, the decline of interior forest area might have con-

siderable effects on some species in Rio Maule-Cobquecura. For instance, the fragmentation of Maulino forests might increase the predation of ground-nesting birds by decreasing the distance of interior habitats at less than 100 m from the edge (Vergara, 2005). Other ecological processes that require an interior habitat defined by a distance greater than 300 m from the edge may have disappeared or severely affected, due to the total loss of these core areas in 1990. In Ohio, USA, the presence of forest core area is critical for preservation of amphibian diversity. In that study, overall salamander diversity was positively associated with the amount of forest within 200 m core zone (Porej et al., 2004).

Observed trends in the measures of the spatial configuration of non-forest cover types were useful to explain some changes in the pattern of forest cover that have occurred since the 1970s. While the area of exotic-species plantation cover increased in aggregation, native forests and shrublands became gradually more fragmented. During the latter stages of deforestation, exotic-species plantations have become the dominant land cover type in the landscape examined, replacing the dominance of native forests, shrublands, and arboreous shrublands observed in the earliest stage and intermixing with the other cover types. Also, in the 1970s native forests were primarily surrounded by arboreous shrublands in the earliest stage of deforestation. Currently, *P. radiata* industrial plantations largely dominate the neighbouring areas of isolated native forest patches. A similar situation has been reported in New South Wales, Australia, where the establishment of *P. radiata* commercial plantations has caused the isolation of remnant eucalypt patches, which has in turn affected the patch occupancy of a small marsupial carnivore (Banks et al., 2005). The progressive fragmentation in Rio Maule-Cobquecura has led to microclimatic changes in the edges that have facilitated the establishment of exotic species towards the interior of the forest fragments in Rio Maule-Cobquecura (Bustamante and Grez, 1995). However, the invasions of *P. radiata* seedlings into the *Nothofagus* forest fragments in the study area tend to occur in the edges of forest fragments, reaching the interior areas in highly disturbed and open forest fragments (Bustamante et al., 2003). Although *P. radiata* seeds are dispersed into the native forests, seedling establishment occurs only at the edges (Bustamante and Simonetti, 2005).

The analysis of spatial patterns of landscape indices needs to be understood as a first step to understanding ecological processes, and not as end itself (Li and Wu, 2004). Due to this, it is highly necessary to complement this work with other surveys that allow a comprehensive assessment of the forest fragmentation at the landscape level.

#### 5.4. Conservation implications

The progressive fragmentation might have significant ecological implications for species dependent on high-quality habitats situated in the interior regions of forest patches. For example, in the lowland Amazonia rain forest the experimental forest fragmentation has resulted in significantly different colonization rates between continuous and fragmented forests (Bruna et al., 2005). In the Atlantic forest landscape, abundance and  $\alpha$  diversity of small mammals were lower in small and medium-sized fragments than in large fragments and

continuous forest, and in isolated compared to connected fragments (Pardini et al., 2005). In the present study area, the division of forest patches is seriously affecting the regeneration of shade-tolerant plants that are dependent on biotic pollinators and dispersers (Bustamante and Castor, 1998; Grez et al., 1998). Also, a study conducted in the Maulino forest revealed that predation of large seeds is higher in small fragments than in continuous forests (Donoso et al., 2003). This may have serious consequences for large-seeded tree species, such as the endangered *Gomortega keule* and *Pitavia punctata*, as small seeded plants, such as many invasive species, may be favoured by forest fragmentation (Donoso et al., 2003). In the same area, the fragmentation has reduced the germination capacity of seeds of *Lapageria rosea*, which may have negative effects on seedling establishment and long-term survival of the species (Henríquez, 2004). Also, some evidence has been found that fragmentation of Maulino forests might significantly reduce reproductive success of ground-nesting birds, such as the endemic rhynocryptids, as a result of the increase in predator abundance in small fragments (Vergara and Simonetti, 2003). Moreover, some results suggest that for south-temperate forest birds, large fragments should afford better protection against extinction than small forest patches (Cornelius et al., 2000). According to all these evidences and the patterns reported in this study, the fragmentation of Maulino forest is leading to substantial changes of the original forest composition at the landscape level. Therefore, the loss of forest habitats and the increasing trend of fragmentation over the next decades may have negative consequences on the flora and fauna existing in the remnant forests, due to changes in composition of assemblages and changes in ecological processes (Forman and Godron, 1986; Bennett, 2003).

Isolation is also an important component of the landscape to assess biological exchange through the landscape. In the Atlantic forest landscape, some understory bird species were significantly affected by the distance between patches (Uezu et al., 2005). The increasing isolation of forest patches in Rio Maule-Cobquecura may also have substantial impacts on some threatened species by reducing the genetic flow between subpopulations of wind-dispersed trees (Bustamante and Castor, 1998).

Maintaining large forest blocks is of importance in the region studied in order to preserve interior species, many of which are listed in the Chilean Red Data Book (Benoit, 1989). As more forests are transformed to younger successional stages in Rio Maule-Cobquecura and are seldom permitted to reach full maturity, species adapted to the ecological patterns and processes that prevailed previously are at a serious disadvantage as evidenced by the large numbers of species at threat of extinction in the region (Bustamante and Castor, 1998; Oldfield et al., 1998; Hechenleitner et al., 2005). The long-term conservation of these species depends on the recovery and sustainability of natural conditions where these species occur, which implies an improvement of integrity and connectivity of the remaining native forest fragments.

Assessment of the impacts of the deforestation and fragmentation recorded here on species diversity and the viability of populations of threatened species are key priorities for future research in the study area. Many researchers suggest that a monitoring programme is highly needed in regions such as this, where fragmentation or habitat changes can negatively affect

biodiversity (Gigord et al., 1999; Jeanneret et al., 2003). These impacts might be monitored on a large scale and over long periods of time using land cover data (Luque, 2000; Honnay et al., 2002).

Similarly, urgent long-term conservation actions are needed in this type of landscape that continue to be intensely used for commercial purposes and where the lack of effective conservation planning has inadequately exposed an area of high diversity and endemism to extreme rates of forest loss and fragmentation. Conservation strategies that encourage the implementation of land use planning in a productive matrix have been proposed as a priority for the conservation of the biodiversity in this type of landscape (Armesto et al., 1998). Additionally, some researchers have emphasised that conservation and land use management strategies should always consider the quality of the whole landscape and especially the number of different habitats and their spatial arrangement (Steiner and Köhler, 2003).

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## 6. Conclusion

This work has succeeded in quantifying the major changes in forest cover in south-central Chile, and in assessing the substantial forest loss that has taken place over the last 25 years in the study area. The research applies standard remote sensing methods in conjunction with spatial pattern analyses to an area of southern-hemisphere temperate forest never previously studied. The analyses conducted illustrated how forest loss was strongly associated with an increase in area of exotic-species plantations. The changes in pattern, in turn, were related to increased forest fragmentation, which was reflected by significant modification of size, density, edge, isolation, connectivity, and core area of forest patches.

The successful description of pattern change accompanying deforestation and forest fragmentation provides a critical component of habitat analysis. At a local level, these changes may result in the elimination, displacement, or enhancement of species populations. Additionally, the identification of these patterns is important to facilitate future landscape management and monitoring actions in this type of forest. Proactive management with a focus on biodiversity conservation and sustainable use is urgent. Although an analysis of the effects of fragmentation on the species and genetic levels of biodiversity was outside the scope of the present study, the description of landscape spatial pattern provides a basis for future research investigating such impacts.

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**Appendix 1. Confusion matrices**

Classified data	Reference data							User accuracy (%)	Error commission (%)		
	Plantation	Shrubland	Arboreus shrubland	Arable land	Water	Native forest	Total				
<b>(a) MSS 1975</b>											
Plantation	25	0	0	0	0	5	30	83.3	0.2		
Shrubland	0	30	0	3	0	4	37	81.1	18.9		
Arboreus shrubland	0	0	31	0	0	4	35	88.6	11.4		
Arable and pasture land	1	1	0	30	2	1	35	85.7	14.3		
Water	0	0	0	1	19	0	20	95.0	5.0		
Native forest	10	13	19	0	0	170	212	80.2	19.8		
Total	36	44	50	34	21	184	369				
Producer's accuracy (%)	69.4	68.2	62.0	88.2	90.5	92.4					
Error omission (%)	30.6	31.8	38.0	11.8	9.5	7.6					
Overall classification accuracy: 82.7%											
<b>(b) TM 1990</b>											
Plantation	28	0	3	1	0	5	37	75.7	0.2		
Shrubland	0	38	2	2	0	10	52	73.1	26.9		
Arboreus shrubland	0	0	44	0	0	11	55	80.0	20.0		
Arable and pasture land	0	1	0	19	0	0	20	95.0	5.0		
Water	0	0	0	1	21	0	22	95.5	4.5		
Native forest	7	5	10	0	0	150	172	87.2	12.8		
Total	35	44	59	23	21	176	358				
Producer's accuracy (%)	80.0	86.4	74.6	82.6	100.0	85.2					
Error omission (%)	20.0	13.6	25.4	17.4	0.0	14.8					
Overall classification accuracy: 83.3%											
Classified data	Reference data								User's accuracy	Error commission(%)	
	Plantation	Shrubland	Arboreus shrubland	Arable land	Water	Native forest	Bare ground	Recent harvest			
<b>(c) ETM+ 2000</b>											
Plantation	25	0	0	0	0	0	0	0	25	100.0	–
Shrubland	0	26	4	2	0	0	0	2	34	76.5	23.5
Arboreus shrubland	3	2	21	0	0	2	1	1	30	70.0	30.0
Arable and pasture land	2	3	1	30	0	1	0	0	37	81.1	18.9
Water	0	0	0	0	25	0	1	0	26	96.2	3.8
Native forest	1	0	2	0	0	51	0	2	56	91.1	8.9
Bare ground	0	0	0	0	2	0	5	0	7	71.4	28.6
Recent harvest*	1	0	0	0	0	1	0	9	11	81.8	18.2
Total	32	31	28	32	27	55	7	14	226		
Producer's accuracy (%)	78.1	83.9	75.0	93.8	92.6	92.7	71.4	64.3			
Error omission (%)	21.9	16.1	25.0	6.3	7.4	7.3	28.6	35.7			
Overall classification accuracy: 84.9%											

\* Recent harvest of commercial plantations (mainly *Pinus radiata*).

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