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Native forest replacement by exotic plantations in southern Chile (1985–2011) and partial compensation by natural regeneration

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ABSTRACT

Although several studies have reported rates of deforestation and spatial patterns of native forest fragmentation, few have focused on the role of natural forest regeneration and exotic tree plantations on landscape dynamics. The objective of this study was to analyze the dynamics of land cover change in order to test the hypothesis that exotic tree plantations have caused a major transformation of temperate forest cover in southern Chile during the last three decades. We used three Landsat satellite images taken in 1985 (TM), 1999 (ETM+), and 2011 (TM) to quantify land cover change, together with a set of landscape indicators to describe the spatial configuration of land cover. Our results showed that the major changes were dynamic conversion among forest, exotic tree plantation and shrubland. During the study period, the area covered by exotic tree plantations increased by 168% (20,896–56,010 ha), at an annual rate of 3.8%, mostly at the expense of native forest and shrubland. There was a total gross loss of native forest of 30% (54,304 ha), but a net loss of initial cover of only 5.1% (9130 ha), at an annual net deforestation rate of 0.2%. The difference between gross and net loss of native forest was mostly the result of conversion of shrubland and agricultural and pasture land to secondary forest following natural regeneration. Over the course of the study period, exotic tree plantations showed a constant increase in patch density, total edge length, nearest-neighbor distance, and largest patch index; maximum mean patch size occurred in the middle of the study period. Native forest exhibited an increase and then a decrease in patch density and total edge length, whereas mean patch size and largest patch index were lowest in the middle of the period. Overall, the observed trends indicate expansion of exotic tree plantations and increase in native forest loss and fragmentation, particularly between 1985 and 1999. Forest loss included both old-growth and secondary forests, while native forest established after secondary succession differed in diversity, structure, and functionality from old-growth and old growth/secondary forests. Since different successional stages influence the provision of ecosystem services, the changes observed in our study are likely to have consequences for humans that extend beyond immediate changes in land use patterns.

1. Introduction

Humans have changed land use and land cover for millenia, resulting in significant impacts on the environment (e.g. Klein Goldewijk et al., 2011). Human activities and demands are rapidly changing ecosystems and landscapes, and only small or remote areas of the globe show no evidence of human intervention (Lambin and Meyfroidt, 2011). Transformation of natural landscapes has eroded ecosystem functions, and habitat loss and fragmentation have increased vulnerability to edge effects and biodiversity loss (Laurance et al., 2006). Acting in the opposite direction is passive regeneration, which may counteract the effects of habitat loss and fragmentation (Morrison and Lindell, 2010; FAO, 2011), especially for areas where natural recolonization is fast due to seed availability, extensive residual cover of natural habitat, and conserved soil (Prach et al., 2007; Chazdon, 2008). Given their complexity, the processes involved in land cover change are the focus of research programs and strategies for sustainable management (Vitousek, 1994). Although important advances have been made, significant gaps remain in our understanding of the spatial ecology of these changes (Iverson et al., 2014).

Among the drivers of land cover change, tree plantations play an important role in many parts of the world. Tree plantations

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are typically established on cleared agricultural land, but they also expand at the expense of native forest, which is an emerging cause of forest loss and fragmentation worldwide (Foley et al., 2005). Three factors have caused the expansion of tree plantations towards increasingly difficult-to-reach areas: depletion of finite resources, particularly timber; natural limits to increasing yield on high-quality land; and development of tree plantation technology feasible on cheaper marginal lands (Kröger, 2013). As a result, planted forests are rapidly expanding worldwide and they currently account for ca. 7% of the total forest area, whereas the area covered by native forests declined by 5.2 million ha annually between 2000 and 2010 (FAO, 2011; Kröger, 2013). The two main areas where plantation expansion has been particularly dramatic are South America, where area increased by 67% between 1990 and 2010, and the Asia–Pacific region, where area increased by 61.6% (Kröger, 2013).

Some tree plantations are intended to provide chiefly environmental benefits, including those fostered by the European Community Agrarian Policy (European Commission, 2013) and the Chinese Grain-to-Green project (Song et al., 2014). However, most tree plantations are grown primarily for producing wood efficiently and for contributing significantly to economic growth; these activities may produce substantial changes in natural ecosystems, with impacts on biodiversity and ecosystem services (Hartley, 2002). Furthermore, management practices such as periodic clearing of understory vegetation can have more drastic effects than any competitive or allopathic effects due to the planted trees (Atauri et al., 2004). The global trend of tree plantation expansion is likely to continue, especially for the production of biofuels (Kole et al., 2012) and carbon storage (Lindenmayer, 2009), while natural forests are in decline and increasingly fragmented (FAO, 2010).

In Chile, tree plantation establishment began in the 1940s (Toro and Gessel, 1999), and in the 1970s the country showed the highest annual rates of plantation increase in South America, especially between 1995 and 2009, due to both afforestation (49,020 ha) and reforestation (53,610 ha) (FAO, 2010; INFOR, 2010). At present, forest plantations are dominated by *Pinus radiata* (D. Don) and *Eucalyptus* spp., which account for 2.3 million ha (INFOR, 2013), an area increasing by 37,000 ha annually (CONAF, 2014). The geographic range of southern temperate forest has declined considerably during the last century (Smith-Ramírez, 2004), partly as a result of conversion of native forest to other land cover types. These processes, together with fragmentation of remnant habitats, threaten native forest in southern Chile (Echeverría et al., 2006; Lara et al., 2011; Nahuelhual et al., 2012). These temperate rainforests are globally important ecoregions because of their biodiversity (Myers et al., 2000; Smith-Ramírez, 2004; Smith-Ramírez et al., 2007), and they have been targeted for urgent conservation by the World Bank, the World Wildlife Fund and other organizations (Dinerstein et al., 1995). In Chile, the last remnants of temperate forest are restricted to upper elevations in the Andean mountains and the southern section of the Coastal Range, where continuous tracts of forest still exist (Smith-Ramírez, 2004).

Studying land cover change has always been limited by data availability. The development of Geographical Information Systems (GIS) has offered a variety of tools for analyzing landscape spatial patterns (Franklin, 2001). The evaluation of temporal forest change based on satellite imagery can then be linked to fragmentation analysis, representing a valuable set of techniques for assessing the severity of threats to ecosystems (Dávalos et al., 2014; Kumar et al., 2014). Many indices have been developed to quantify patterns at the landscape scale, including area (Armenteras et al., 2003), edge (Franklin, 2001; McGarigal et al., 2012), shape (Franklin, 2001; Mcgarigal et al., 2012), distance (Mcgarigal et al., 2012), and connectivity metrics (Franklin, 2001; Mcgarigal et al., 2012). The use of these metrics in deforestation and fragmentation studies has increased exponentially around the world in recent decades (Willson et al., 1994; Armenteras et al., 2003; Cayuela et al., 2006a,b; Dávalos et al., 2014), probably motivated by increasing accessibility to remote sensing data and powerful computers (Newton et al., 2009).

Relatively few studies have analyzed land cover change and forest fragmentation in Chile (Echeverría et al., 2006; Schulz et al., 2010; Nahuelhual et al., 2012; Altamirano et al., 2013). Echeverría et al. (2006) assessed the patterns of deforestation and forest fragmentation in the Coastal Range of south-central Chile over a 25-year period using data from 1975, 1990 and 2000. Schulz et al. (2010) investigated land cover changes and major trends in landscape dynamics in central Chile, including regeneration, using multi-temporal satellite images taken in 1975, 1985, 1999 and 2008. Nahuelhual et al. (2012) analyzed the drivers of plantation expansion in south-central Chile for the periods 1975–1990 and 1990–2007. Finally, Altamirano et al. (2013) analyzed patterns of deforestation and fragmentation in south-central Chile using fine-resolution (0.0225 ha) classified maps from satellite images taken in 1986, 1999 and 2008.

Notwithstanding the growing literature on land cover change, few studies have investigated simultaneously how landscape dynamics are affected by natural forest regeneration (e.g. Pütz et al., 2011; Schulz et al., 2010) and exotic tree plantations (e.g. Nahuelhual et al., 2012). Improving our understanding of such dynamics may help mitigate or reverse their impact on forest ecosystems, contribute to land use planning, and guide the design and implementation of conservation and restoration programs at the landscape scale. The objective of this study was to analyze the dynamics of land cover change in order to test the hypothesis that exotic tree plantations have caused a major transformation of temperate forest cover in southern Chile in a recent time period spanning 26 years. To achieve this goal, our study attempted to (1) determine the rates and amount of land cover change, (2) measure the spatial distribution of forest loss and expansion of plantations, and (3) examine whether natural regeneration has compensated for forest loss at the landscape scale.

2. Methods

2.1. Study area

The study area covers ca. 2700 km^2 of the Chilean Coastal Range (Fig. 1), including rivers and wetlands, and elevation ranges from 4 to 684 m. It has abundant endemic flora and fauna, which reflect the location of vegetation refuges during the last glacial period (Armesto et al., 1995). Evergreen forests are the dominant vegetation type, occupying 79% of the total forest cover in the study area (CONAF-CONAMA-BIRF, 1999). The predominant climate is temperate with Mediterranean influence, a mean annual temperature of 11 °C and a mean annual precipitation of 2500 mm. Soils are derived from metamorphic material and granitic rocks (IREN-CORFO, 1964).

Land tenure is characterized by a mosaic of different land cover types, productive activities, and local stakeholders. The dominant types of land tenure correspond to properties owned by forest companies (81,100 ha, 30% of the study area) that concentrate the area covered by exotic tree plantations, private protected areas (52,000 ha, 19.3%), and small properties (46,827 ha, 17%) owned by "campesinos". This Spanish name refers to rural people in the subsistence economy who live on <200 ha, as defined in Chilean law. The other major types of land tenure are large properties, i.e. $\geqslant$200 ha (45,663 ha, 16.97%) and public protected areas (26,000 ha, 9.74%). Most campesino-owned small properties show frequent and intense alterations due to constant efforts to achieve

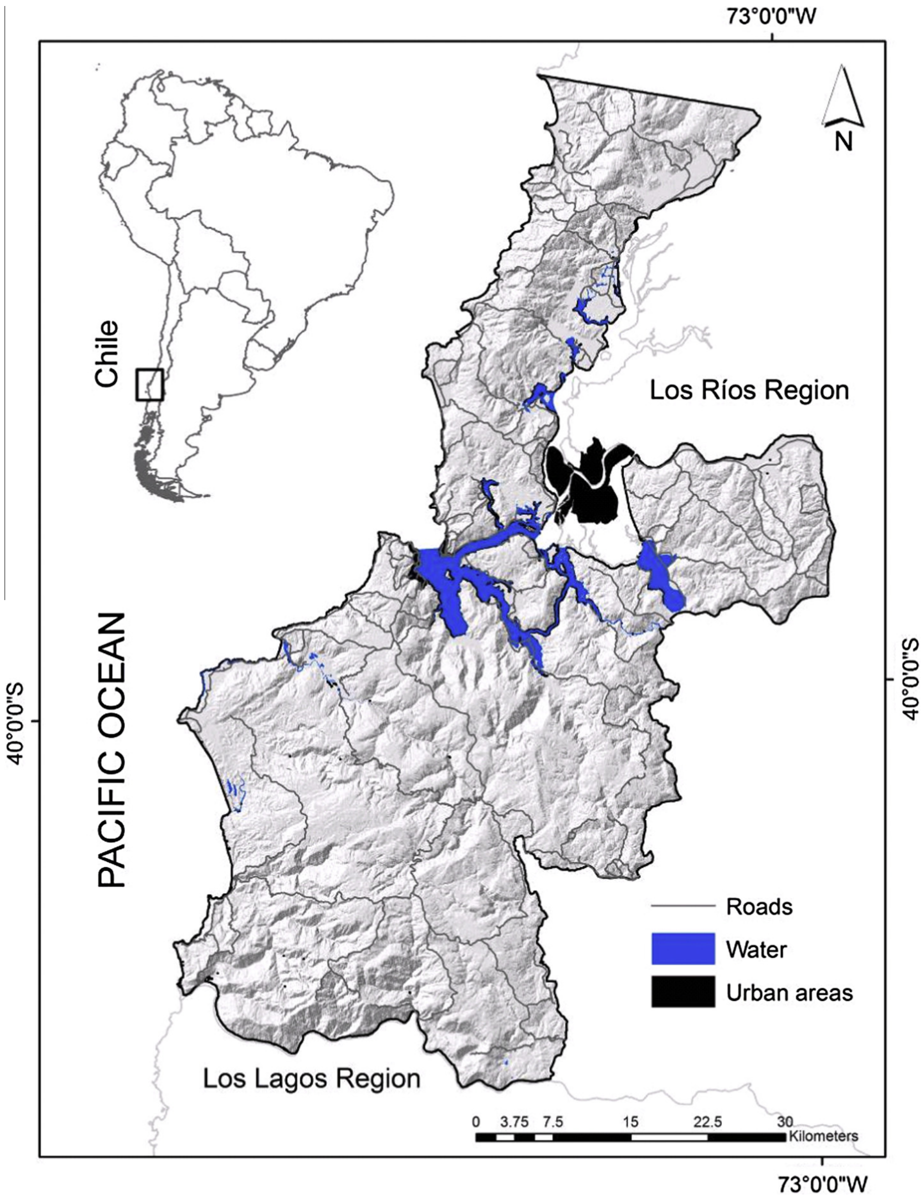

Fig. 1. Location of the study area within the Coastal Range of southern Chile.

adequate production for family subsistence (Zamorano-Elgueta et al., 2012).

2.2. Image classification

A set of three Landsat images (path 233, row 88) were acquired for the years 1985 (TM), 1999 (ETM+), and 2011 (TM), with a pixel spatial resolution of 30×30 m. All images were taken during the dry season (December to February). The images were pre-processed, including geometric, atmospheric and topographic corrections (Hantson and Chuvieco, 2011). We defined five classes of land cover: (1) native forest, (2) exotic tree plantation, (3) shrubland, (4) agricultural and pasture land, and (5) bare ground. Exotic tree plantations corresponded to industrial plantations of exotic tree species dominated by *P. radiata* (D. Don) and *Eucalyptus* spp. (Reyes and Nelson, 2014).

Landscape complexity poses particular challenges for image classification (Cayuela et al., 2006a). Frequent misclassification is inevitable, particularly if various categories are interspersed within a small spatial area (Foody, 2002) or if some of the land cover categories have overlapping spectral signatures (Pedroni, 2003). For example, vegetation stages following successional gradients, such as shrubland, arboreous-shrubland and forest categories (Echeverría et al., 2006), usually show very similar spectral signatures, as do forest successional stages, such as young, intermediate and old forest (Liu et al., 2008). For this reason, we included within the native forest category the three main forest successional stages defined by the Chilean native forest cadastre (CONAF-CONAMA-BIRF, 1999): old-growth, old-growth/secondary, and secondary forest.

To classify the scenes we used a supervised classification method. Training sites were selected to represent the spectral variability of each land cover class and were extracted from color composite images and based on local knowledge (Chuvieco, 2010). The number of training sites for the various land cover classes for the three images studied ranged between 18–32, 19–35, and 18–32. The maximum likelihood algorithm was used to assign probabilities of membership to each class and each pixel was assigned to the most probable class (Richards and Jia, 2006).

2.3. Accuracy assessment

The Bhattacharyya distance was used as a statistical measure of between-class separability based on spectral signatures (Bhattacharrya, 1943). One advantage of this index is its close relationship to the probability of accurate classification (Choi and Lee, 2001). A separability value of 2.0 indicates a proper pixel separation with no more pixel overlap. Good separability is attained with values between 1.9 and 2.0. Values between 1.9 and 1 indicate that two classes can be separable to some extent, whereas values less than 1 indicate very poor separability. The equation we used to compute the Bhattacharryya distance is described in the PCI user manual (PCI, 2001).

Accuracy of the 1999 and 2011 scenes was assessed using ALOS scenes from 2010 with a pixel spatial resolution of 10×10 m, and the most updated version of the cadastre for the Región de los Ríos (CONAF-CONAMA, 2008). The cadastre was developed at the 1:50,000 scale, and was derived from aerial photographs and satellite imagery. To classify the 1985 scene, we used forest cover maps generated from aerial photographs of 1985 at a scale of 1:60,000. We also used two maps of land cover from 1986 and 1999 derived from Landsat scenes to assess the accuracy of the 1985 and 1999 classifications (González et al., 2005). Sets of 173 and 199 control points were used for the 1985 and 1999 scenes, respectively. The points were overlaid onto the reference land cover maps and assigned to respective classes. In order to assess the accuracy of the 2011 image, 198 ground control points were visited in the field during the dry season in 2012. Confusion matrices were constructed to cross-validate the land covers derived from the satellite scenes. Three accuracy measures were calculated: producer's accuracy, user's accuracy, and overall accuracy. Most processing was performed using PCI 7.0 (PCI, 2001) and ArcGis 10 (ESRI, 2011). Finally, we compared the overall area estimated for each land cover type in our 1985, 1999 and 2011 classifications with estimates obtained from technical reports (González et al., 2005; TNC, 2011) and regional statistics (CONAF-CONAMA-BIRF, 1999; INFOR, 1986).

2.4. Cover change and spatial configuration of tree plantations and native forest

To compare the change in cover of exotic tree plantations and native forest for the 1985–1999 and for 1999–2011 periods, we used the compound interest rate formula proposed by Puyravaud (2003):

$$Change\,rate = 100/(t_2 - t_1)x\,ln(A_2/A_1)$$

where A_1 and A_2 are the cover of exotic tree plantations or native forest at times t_1 and t_2. Both net and gross changes were calculated. Net change represents the difference between the gains and losses in a cover type between two periods. Gross change represents the total area modified between the two periods. The spatial configuration of fragments was quantified and compared using the following landscape metrics: (a) patch density (number of patches/100 ha), (b) total edge length (km), (c) mean patch size (ha), (d) Euclidean nearest-neighbor distance (m), and (e) largest patch index (%). Landscape spatial indices were computed using FRAGSTATS (version 4.2, Mcgarigal et al., 2012).

3. Results

3.1. Accuracy assessment

The average between-class separability based on the Bhattacharrya distance ranged from 1.948 to 1.997. Signature separability between exotic tree plantation and native forest was 1.987, 1.999, and 1.991 for the 1985, 1999, and 2011 scenes, respectively, indicating good spectral separability between these two land cover classes.

Overall accuracy for classification was 73.2%, 83.9%, and 82.9% for the 1985 TM, 1999 ETM+, and 2011 TM scenes, respectively. The lowest producer's accuracy was obtained in exotic tree plantations for the 1985 and 1999 scenes, and in shrublands for the 2011 scene, whereas the lowest user's accuracy was obtained in exotic tree plantations for the 1985 scene, and in shrublands for the 1999 and 2011 scenes (Appendix A). Classification accuracy was intermediate for exotic tree plantations; nevertheless, overall area estimates for 1985 (20,896 ha) and 1999 (44,921 ha) were roughly similar to estimates from other sources: 15,000–20,000 ha for 1985 and 40,000–46,000 ha for 1999 (INFOR, 1986; CONAF-CONAMA-BIRF, 1999; González et al., 2005; TNC, 2011).

3.2. Changes in land cover

All land cover classes except exotic tree plantations showed a net loss (Table 1). Net losses were highest for shrubland (18,906 ha, −7.4% of the study area), followed by native forest (9130 ha, −3.6%) and agricultural and pasture land (7263 ha, −2.8%). Conversely, exotic tree plantations gained 35,114 ha or 13.7%. These changes were more intense between 1985 and 1999 than between 1999 and 2011 (Fig. 2, Appendix B).

Over the study period, the dominant land cover class was native forest, which decreased from 69.9% of the study area (179,663 ha) in 1985 to 66.4% (170,534 ha) in 2011; it was 66.5% (170,743 ha) in 1999. Between 1999 and 2011, total native forest cover remained relatively stable (Fig. 2, Appendix B). The annual net deforestation rate was 0.2% (351 ha per year) over the entire study period, and higher in 1985–1999 (0.36% or 637 ha) than in 1999–2011 (0.01% or 17 ha per year). Native forest was distributed across the entire study area but concentrated in the southern parts, where it showed a continuous distribution (Fig. 2). In the central and eastern parts of the study area, native forest occurred as a higher number of smaller patches.

Exotic tree plantations represented 8.1% of the study area (20,896 ha) in 1985, 17.5% (44,921 ha) in 1999, and 21.8% (56,010 ha) in 2011 (Fig. 2, Appendix B). In other words, exotic tree

Table 1
Net change (gain minus loss) for land cover classes in hectares and as a percentage of the study area.

Cover type	1985–1999						1999–2011					
	Gains		Losses		Net change		Gains		Losses		Net change	
	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%
Native forest	21,987	8.5	30,908	−12.0	−8,921	−3.5	23,188	9.0	23,397	−9.1	−209	−0.1
Exotic tree plantation	30,245	11.8	6219	−2.4	24,025	9.4	21,125	8.2	10,036	−3.9	11,089	4.3
Shrubland	12,346	4.8	23,790	−9.3	−11,444	−4.5	7261	2.8	14,723	−5.7	−7462	−2.9
Agricultural and pasture land	7712	3.0	13,710	−5.3	−6000	−2.3	7122	2.7	8385	−3.3	−1263	−0.5
Bare ground	3114	1.2	776	0.3	2338	0.9	414	0.2	2570	−1.0	−2155	−0.8

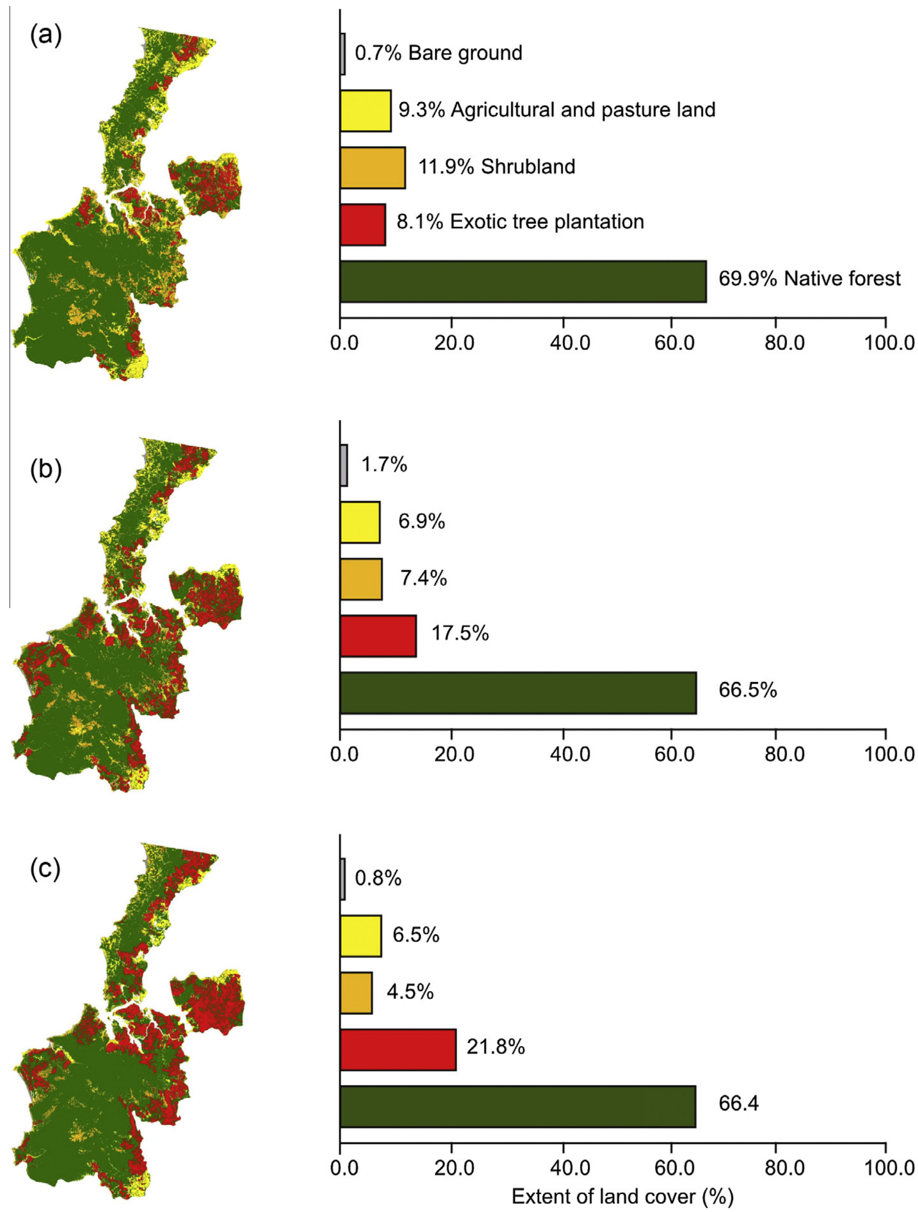

Fig. 2. Land cover maps based on the classification of TM and ETM + Landsat images for the years (a) 1985, (b) 1999, and (c) 2011, and comparison of the extents of land cover classes as percentages of the study area.

plantations increased by 168% from 1985 to 2011, with an annual net gain of 3.8% equivalent to 1351 ha per year. This rate was higher in 1985–1999 (1717 ha/yr, 5.5%) than in 1999–2011 (924 ha/yr, 1.8%). In 1985, exotic tree plantations were concentrated in the eastern and central parts of the study area, whereas in 1999 they extended over the entire study area, especially in the northern and central parts (Fig. 2). This expansion continued in 2011, but at a lower rate.

Shrubland and agricultural and pasture land represented 11.9% (30,461 ha) and 9.3% (23,911 ha) of the study area, respectively, in 1985, 7.4% (19,017 ha) and 6.9% (17,913 ha) in 1999, and 4.5% (11,555 ha) and 6.5% (16,650 ha) in 2011 (Fig. 2, Appendix B).

3.3. Change trajectories among land cover classes

The major changes occurred among native forest, shrubland, and exotic tree plantations and, to a lesser extent, between these three types of land cover classes and agricultural and pasture land (Fig. 3). Between 1985 and 1999, changes consisted mainly of conversion of native forest to exotic tree plantations (7.4% of the study area) and to shrubland (3.3%). Also remarkable during this period was the conversion of shrubland to native forest (4.4% of the study area), exotic tree plantations (2.7%), and agricultural and pasture land (1.8%), as well as conversion of agricultural and pasture land to native forest (2.1%) and exotic tree plantations (1.6%) (Fig. 3).

Between 1999 and 2011, the major changes were the conversion of native forest to exotic tree plantations (6.5% of the study area), shrubland (1.4%), and agricultural and pasture land (1.2%). Agricultural and pasture land changed mainly to native forest (1.6%), whereas shrubland changed to native forest (3.9%), agricultural and pasture land (0.9%), and exotic tree plantations (0.8%) (Fig. 3).

3.4. Changes in spatial configuration

Exotic tree plantations showed a constant increase in patch density (Fig. 4a), total edge length (Fig. 4b), nearest-neighbor

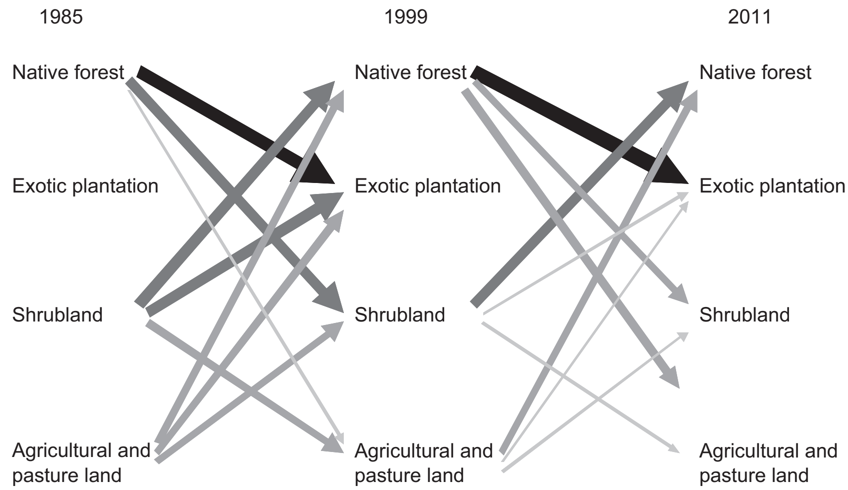

Fig. 3. Major change trajectories and their contributions to net change among land cover classes in the study area. Thick arrows correspond to net change of > 6% of the study area, intermediate arrows to net change of 2.5–4.4%, and thin arrows to net change of 1–1.9%. The thinner arrows correspond to a marginal net change of < 1%.

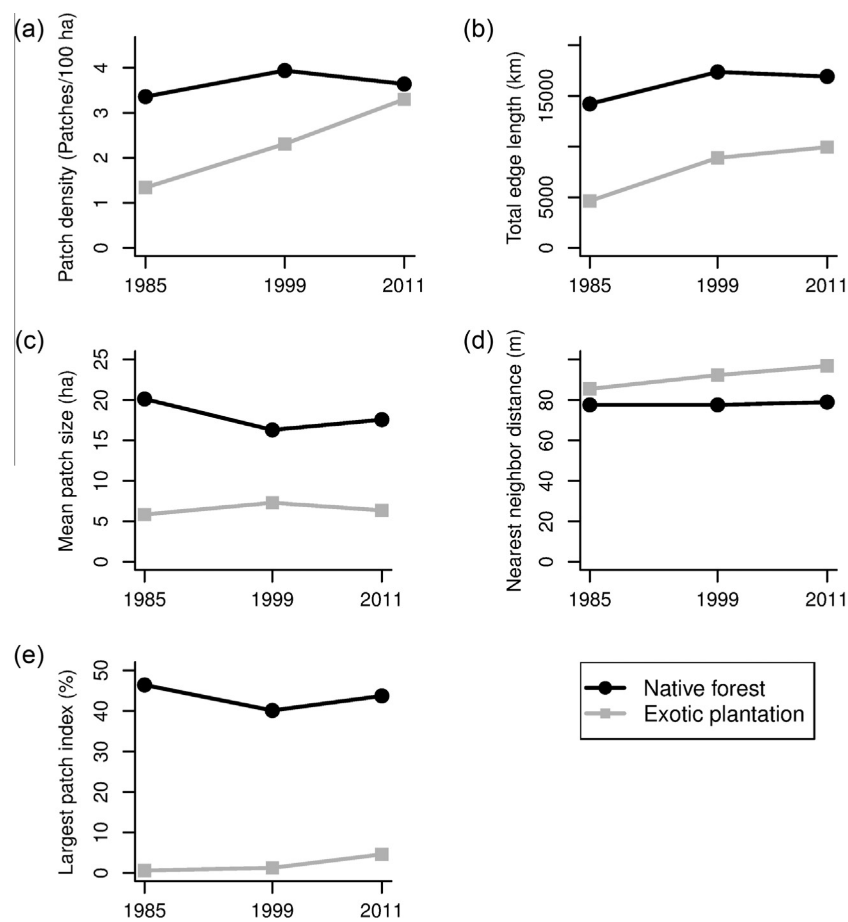

Fig. 4. Variation in landscape metrics of exotic tree plantation and native forest fragments for the years 1985, 1999, and 2011: (a) patch density, (b) total edge, (c) mean patch size, (d) nearest-neighbor distance, and (e) largest patch index.

distance (Fig. 4d), and largest patch index (Fig. 4e) along the studied years, with greatest mean patch size occurring in the middle year (Fig. 4c). Native forest exhibited an increase and then a decrease in patch density (Fig. 4a) and total edge length (Fig. 4b), whereas mean patch size (Fig. 4c) and largest patch index (Fig. 4e) were lowest in the middle year. The nearest-neighbor distance showed minimal variation over the study period (Fig. 4d).

4. Discussion

4.1. Evaluation of methods and results

Monitoring of land cover change based on remote sensing data is certainly an imprecise task (Foody, 2002). Classifications of satellite imagery into land cover types are never completely accurate, which affects analyses of forest loss and landscape patterns (Cayuela et al., 2006a,b; Echeverría et al., 2006). According to the confusion matrices (Appendix A), image accuracy tended to improve as the image date became more recent. This might be related to the availability of concurrent land cover maps and field data that supplement the more recent images.

The high overall accuracy of the 1999 and 2011 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. The classification conducted for the oldest scene (TM 1985) suffered from the disadvantage of limited ground validation and lower values of accuracy, particularly for exotic plantations. In contrast, the estimated area of land cover classes derived from the 1985 and 1999 classifications matched well estimates derived from alternative sources (INFOR, 1986; González et al., 2005; CONAF-CONAMA-BIRF, 1999; TNC, 2011). This indicates that our analysis can accurately reflect coarse-grained landscape patterns, with uncertainty increasing as we narrow down our focus to pixel grain.

4.2. Land cover changes

Our study points to three major, concurrent changes in southern Chilean temperate forests in recent decades: (1) intense expansion of exotic tree plantations, (2) reduction and fragmentation of native forest and shrubland and (3) natural regeneration of native forest at the expense of shrubland and agricultural and pasture land. Exotic tree plantation area increased the fastest of all land cover classes, as reported by Echeverría et al. (2006). Exotic tree plantations increased from 8% of the study area in 1985 to over 20% in 2011. This rate of increase is consistent with results from the adjacent region in the Coastal Range of south-central Chile: Echeverría et al. (2006) measured an annual deforestation rate of 4.5% for the period 1975–2000, and Nahuelhual et al. (2012) highlighted the rapid expansion of exotic tree plantations from 5.5% to 42.4% of the landscape, at annual rates of 7.9% for 1975–1990 and 5.1% for 1990–2007. Conversely, native forest initial cover showed a net loss of only 5.1% over the study period, at an annual deforestation rate of 0.2%. This discrepancy between gross and net forest loss is mostly the result of conversion of shrubland and agriculture and pasture land to secondary forest.

The expansion of exotic tree plantations may have several negative impacts on the ecological functioning of the landscape. Exotic forest plantations are widely thought to be less favorable habitats than native forests (Carnus et al. 2006). Comparisons of unmanaged forests and exotic tree plantations have shown that exotic plantations contain impoverished flora (Hartley, 2002) and fauna (Schnell et al., 2003). Plantations may be unsuitable for many native species because of the loss of some structural components of native habitats, such as understory vegetation (Brockerhoff et al., 2008), which are critical for some wildlife species. The effect of tree plantations on biodiversity depends on the type of plantation and the natural structure of surrounding native forests (Hartley, 2002); in fact, plantations can contribute to biodiversity conservation if correctly designed and managed (Hartley, 2002; Carnus et al. 2006). On the other hand, management of exotic tree plantations, which mainly involves extensive clear cutting, leaves the soil unprotected for several years until the new plantation is established (Pérez, 1999). Exotic tree species, particularly numerous pine species, often invade nearby natural habitat (Williams and Wardle, 2005), which can harm areas set aside for conservation or water production (Kröger, 2013).

A large part of the study area contains exotic tree plantations, which have been identified in other regions of Chile as a direct cause of deforestation and biodiversity loss (Nahuelhual et al., 2012). In addition, much of our study region has been degraded as a result of logging for firewood and clearing for livestock and cultivation (Smith-Ramírez, 2004; Zamorano-Elgueta et al., 2014). Logging and clearing became more intense in central-south and central Chile with European colonization, especially in the Coastal Range (Camus, 2006). In these areas, anthropogenic disturbance poses a serious threat to biodiversity conservation, mainly due to the concentration of the human population and the characteristics of the coastal mountains. These mountains, unlike the Andes Range, spread out in the east–west direction and altitudes do not generally exceed 1500 m, making them more accessible to humans and thereby rendering the forests more vulnerable to threats (Armesto et al., 1995). These features help explain the major expansion of exotic tree plantations in the region (Echeverría et al., 2006; Nahuelhual et al., 2012).

Our results show that land cover change in the Coastal Range of Southern Chile took place as a progressive conversion among forest, exotic tree plantation, shrubland and agricultural and pasture land cover. These patterns are similar to those reported in dryland forest of central Chile, especially in the coastal zone, where frequent exchanges were reported between pasture and shrubland as well as among pasture, bare ground and agricultural areas (Schulz et al., 2010). However, the expansion of exotic tree plantations in central Chile did not result in major conversion of native forest. Instead, forest loss was triggered by the conversion of forest to shrubland, mostly driven by a continuous degradation due to permanent grazing pressure, firewood extraction and charcoal production (Armesto et al., 2007). In general, successional recovery of dry forests is largely constrained by such factors as water availability, soil erosion, human-induced fires and the lower regeneration ability of forest species than shrubland species (Armesto et al., 2007). In our study area, however, these constraints are less severe and do not impede natural regeneration (Albornoz et al., 2013). This is evidenced by the large proportion of shrubland converted to native forest during the study period, in contrast to deforestation patterns in other regions (Vogelmann et al., 2012; Armenteras et al., 2013).

Native forest regeneration may have been facilitated by the creation in 2002 of a 50,000-ha private protected area in the region, which was established in an area historically altered by firewood production, intensive logging of high-value species such as the conifer *Fitzroya cupressoides* (González, 2004) and plantation of exotic trees. In this area, the largest replacement of native forest by exotic tree species in Chile took place during the 1980s and 1990s. After the creation of the protected area, these productive practices were stopped, allowing forest conservation and restoration to be promoted. At present, this area is characterized by abundant forest regeneration, especially seedlings, probably as a consequence of the creation of the private protected area. Furthermore, even though regeneration comprised an age range from recently established seedlings to saplings potentially older than 40 years, most of the regeneration corresponded to seedlings <0.3 m (C. Zamorano-Elgueta, unpublished data). This corresponds to recently established regeneration (<5–10 years old; see Vita, 1977; Donoso et al., 2006). In addition, the two major Chilean forest companies, recently certified by the Forest Stewardship Council (FSC), have started several initiatives of forest conservation and restoration to fulfill the commitments stipulated in this certification. Restoration activities on these companies' properties may spread native forest in the future. Despite these promising trends, replacement of native forests by exotic tree plantations remains a common practice, as indicated in the present study and in previous work (Echeverría et al., 2006; Schulz et al., 2010; Nahuelhual et al., 2012).

Native forest and shrubland were found to be more dynamic, showing larger gains and losses over the study period, than other land cover classes. We observed exchanges between native forest and shrubland as well as between shrubland and agricultural and pasture land, which took place in small patches scattered throughout the study area. Such exchanges resulted in a net loss of native forest and shrubland due to the expansion of exotic tree plantations. This pattern of change is still motivated by the afforestation policies implemented by the Chilean government since the 1970s to promote fast-growth tree plantations; these policies include subsidies covering 75–90% of afforestation costs (Reyes and Nelson, 2014).

4.3. Spatial configuration of changes

Our analysis showed a constant increase in patch density, largest patch index, total edge length and nearest-neighbor distance on exotic tree plantations over the study period, whereas mean patch size increased initially and then decreased. These results indicate that plantations expanded as both continuous and non-continuous patches throughout the study area; in other words, new plantation area appeared both as isolated patches and as patches close to existing plantations. These metrics also initially increased for native forests and then later decreased. The greatest decline in the largest native forest patch size coincided with the time when annual forest loss was greatest, as reported elsewhere (Cayuela et al., 2006b; Echeverría et al., 2006; Schulz et al., 2010). Echeverría et al. (2006) suggested that the constant action of deforestation led to a decline in patch density of native forest in southern Chile. The decline in patch density and other metrics in our study area may be the result of passive conversion of shrubland to native forest and the decreasing annual rate of exotic tree plantation.

Trends in the spatial configuration of exotic tree plantations and native forest may help explain some changes in the pattern of these cover classes since the 1980s. While exotic tree plantations initially increased during the study period, native forest gradually became more fragmented. In 1999, exotic tree plantations became the second dominant land cover class, and replaced the dominance of shrubland and agricultural and pasture land observed in 1985. Native forest was surrounded primarily by shrubland at the beginning of the study period, whereas exotic tree plantations came to dominate the neighboring areas of native forest patches by the end of the period.

4.4. Application to sustainable land use planning

Deforestation and native forest fragmentation in the Coastal Range of Región de Los Ríos was found to be less intensive than in other regions of Chile (Echeverría et al., 2006; Schulz et al., 2010; Nahuelhual et al., 2012). In addition, we found a relatively high rate of passive conversion of shrubland to native forest. These results may reflect the better conservation status of the study area compared to other regions of Chile. Nevertheless, the continuing expansion of exotic tree plantations and loss and fragmentation of native forest may lead to microclimatic changes at the forest edges that may facilitate the spread of exotic species towards the interior of the forest fragments (Murcia, 1995). Whereas forest loss included both old-growth and secondary forests, native forest established after secondary succession differed in diversity, structure, and functionality from old-growth and old growth/secondary forests (Lu et al., 2003), and they respond differently to human impacts in the study area (Zamorano-Elgueta et al., 2014). Different successional stages also provide different levels of ecosystem services including those related to soil (Moran et al., 2000) and water (Lara et al., 2009), in particular during the dry summer season. Thus, these alterations will have consequences for humans beyond the immediate changes in land use patterns (Turner et al., 1993).

Our study is a first step to understanding ecological processes underpinning forest changes in southern Chile, and it is not an end in itself (Li and Wu, 2004). An important complement to the work described here would be to precisely define the native forest categories as old-growth, old-growth/secondary and secondary forest, which show important differences despite their similar spectral signatures. This may improve our understanding of how land cover change and exotic tree plantations influence landscape ecology, including in areas where the percentage of native forest cover shows constant, albeit slight, increases. These results may support conservation or restoration strategies, including the definition of priority areas for conservation or restoration actions. Identifying priority areas would increase the efficiency and impact of available resources to design, plan and establish forest restoration programs, where interventions will produce the greatest benefits, such as in maintaining and enhancing biodiversity and providing ecosystem services. Future research should focus on the management of external influences on forests, such as the expansion and ecological impacts of exotic tree plantations.

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Appendix A

Confusion matrices for Dempster-Shafer classifications of (a) 1985, (b) 1999, and (c) 2011 Landsat scenes. NF, native forests; EP, exotic tree plantation; SHR, shrubland; APL, agricultural and pasture land; BG, bare ground.

Land cover map	Ground verification points 1985 TM						
	NF	EP	SHR	APL	BG	Total	User's accuracy
(a)							
NF	73	8	9	8	0	98	74.5
EP	8	15	0	0	0	23	65.2
SHR	7	0	20	1	0	28	71.4
APL	3	1	2	18	0	24	75.0
BG	0	0	0	0	0	0	0.0
Total	91	24	31	27	0	173	
Producer's accuracy	80.2	62.5	64.5	66.7	0.0		
Overall classification accuracy: 73.2%							
	1999 ETM+						

(continued on next page)

Appendix A (*continued*)

Land cover map	Ground verification points 1985 TM						
	NF	EP	SHR	APL	BG	Total	User's accuracy
(b)							
NF	91	13	0	2	1	107	85.1
EP	8	38	1	0	0	47	80.8
SHR	1	1	18	2	1	23	78.3
APL	1	0	2	16	0	19	84.2
BG	0	0	0	0	3	3	100.0
Total	101	52	21	20	5	199	
Producer's accuracy	90.1	73.1	85.7	80.0	60.0		
Overall classification accuracy: 83.9%							
	2011 TM						
(c)							
NF	89	14	4	1	0	108	82.4
EP	11	50	0	0	0	61	81.9
SHR	1	1	10	1	0	13	76.9
APL	2	0	0	12	1	15	80.0
BG	0	0	0	0	1	1	100.0
Total	103	65	14	14	2	198	
Producer's accuracy	86.4	76.9	71.4	85.7	50		
Overall classification accuracy: 82.9%							

Appendix B

Transition matrices for different land cover changes in southern Chile for the periods (a) 1985–1999 and (b) 1999–2011. NF, native forest; EP, exotic tree plantation; SHR, shrubland; APL, agricultural and pasture land; BG, bare ground.

	NF		EP		SHR		APL		BG		Total 1985	
	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%
(a)												
NF	148,756	57.9	19,035	7.4	8562	3.3	2648	1.0	662	0.3	179,663	69.9
EP	5175	2.0	14,676	5.7	711	0.3	250	0.1	83	0.03	20,896	8.1
SHR	11,250	4.4	6880	2.7	6671	2.6	4652	1.8	1008	0.4	30,461	11.9
APL	5335	2.1	4187	1.6	2826	1.1	10,201	3.9	1361	0.5	23,911	9.3
BG	227	0.1	142	0.06	246	0.1	161	0.06	1153	0.4	1930	0.7
Total 1999	170,743	66.5	44,921	17.5	19,017	7.4	17,913	6.9	4268	1.7	256,861	100.0
	NF		EP		SHR		APL		BG		Total 1999	
	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%
(b)												
NF	147,346	57.4	16,712	6.5	3538	1.4	3073	1.2	74	0.03	170,743	66.5
EP	8352	3.2	34,885	13.6	872	0.3	762	0.3	50	0.02	44,921	17.5
SHR	10,024	3.9	2119	0.8	4294	1.7	2440	0.9	140	0.05	19,017	7.4
APL	4106	1.6	1860	0.7	2268	0.9	9528	3.7	151	0.06	17,913	6.9
BG	706	0.3	434	0.2	583	0.2	847	0.3	1698	0.7	4268	1.7
Total 2011	170,534	66.4	56,010	21.8	11,555	4.5	16,650	6.5	2113	0.8	256,861	100.0

References

Albornoz, F.E., Gaxiola, A., Seaman, B.J., Pugnaire, F.I., Armesto, J.J., 2013. Nucleation-driven regeneration promotes post-fire recovery in a Chilean temperate forest. Plant Ecol. 214, 765–776.

Altamirano, A., Aplin, P., Miranda, A., Cayuela, L., Algar, A.C., Field, R., 2013. High rates of forest loss and turnover obscured by classical landscape measures. Appl. Geogr. 40, 199–211.

Armenteras, D., Gast, F., Villareal, H., 2003. Andean forest fragmentation and the representativeness of protected natural areas in the eastern Andes, Colombia. Biol. Conserv. 113, 245–256.

Armenteras, D., Rodríguez, N., Retana, J., 2013. Landscape dynamics in northwestern Amazonia: an assessment of pastures, fire and illicit crops as drivers of tropical deforestation. PLoS One 8 (1), e54310.

Armesto, J.J., Aravena, J.C., Villagrán, C., Pérez, C., Parker, G., 1995. Bosques templados de la cordillera de la costa. In: Armesto, J., Villagrán, C., Arroyo, M.K. (Eds.), Ecología de los Bosques Nativos de Chile. Editorial Universitaria, Santiago, Chile, pp. 199–213.

Armesto, J.J., Arroyo, K., Mary, T., Hinojosa, L.F., 2007. The Mediterranean environment of Central Chile. In: Veblen, T.T., Young, K.R., Orme, A.R. (Eds.), The Physical Geography of South America. Oxford University Press, New York, pp. 184–199.

Atauri, J.A., De Pablo, C.L., De Agar, P.M., Schmitz, M.F., Pineda, F.D., 2004. Effects of management on understory diversity in the forest ecosystems of Northern Spain. Environ. Manage. 34, 819–828.

Bhattacharyya, A., 1943. On a measure of divergence between two statistical populations defined by their probability distributions. Bull. Calcutta Math. Soc. 35, 99–109.

Brockerhoff, E.G., Jactel, H., Parrotta, J.A., Quine, C.P., Sayer, J., 2008. Plantation forests and biodiversity: oxymoron or opportunity? Biodiversity Conserv. 17, 925–951.

Camus, P., 2006. Ambiente, bosques y gestión forestal en Chile. 1541–2005. Lom ediciones, Santiago, Chile.

Carnus, J.M., Parrotta, J., Brockerhoff, E.G., Arbez, M., Jactel, H., Kremer, A., Lamb, D., O'Hara, K., Walters, B., 2006. Planted forests and biodiversity. J. Forest 104, 65–77.

Cayuela, L., Golicher, J.D., Salas Rey, J., Rey Benayas, J.M., 2006a. Classification of a complex landscape using Dempster–Shafer theory of evidence. Int. J. Remote Sens. 27 (10), 1951–1971.

Cayuela, L., Rey Benayas, J.M., Echeverría, C., 2006b. Clearence and fragmentation of tropical montane forests in the highlands of Chiapas, Mexico (1975–2000). Forest Ecol. Manage. 226, 208–218.

Chazdon, R.L., 2008. Beyond deforestation: restoring forests and ecosystem services on degraded lands. Science 320, 1458–1460.

Choi, E., Lee, Ch., 2001, Estimation of classification error based on the Bhattacharyya distance for multimodal data. Geosci. Remote Sens. Symp.

Chuvieco, E., 2010. Teledetección Ambiental. Editorial Ariel, Barcelona.

CONAF 2014. Estadísticas forestales. <http://www.conaf.cl/nuestrosbosques/bosques-en-chile/estadisticas-forestales/>.

CONAF-CONAMA, 2008. Catastro de uso del suelo y vegetación: Monitoreo y actualización. Región de Los Ríos. Ministerio de Agricultura. Santiago, Chile.
CONAF-CONAMA-BIRF, 1999. Catastro y evaluación de recursos vegetacionales nativos de Chile. Ministerio de Agricultura. Santiago, Chile.
Dávalos, L.M., Holmes, J.S., Rodríguez, N., Armenteras, D., 2014. Demand for beef is unrelated to pasture expansion in northwestern Amazonia. Biol. Conserv. 170, 64–73.
Dinerstein, E., Olson, D., Graham, D., Webster, A., Primm, S., Bookbinder, M., Ledec, G., 1995. A conservation assessment of the terrestrial ecoregions of Latin America and the Caribbean. WWF, World Bank.
Donoso, C., Alarcón, D., Donoso, P., Escobar, B., Zúñiga, A., 2006. Laurelia (=Laureliopsis) philippiana Looser. Tepa, Huahuan, Laurela. Familia: Monimiaceae. In: Donoso, C., (Ed.). Las especies arbóreas de los bosques templados de Chile y Argentina. Autoecología Marisa Cuneo ediciones, Valdivia, Chile, pp. 302–313.
Echeverría, C., Coomes, D., Salas, J., Rey Benayas, J.M., Lara, A., Newton, A., 2006. Rapid deforestation and fragmentation of Chilean temperate forests. Biol. Conserv. 130, 481–494.
ESRI, 2011. ArcGIS Desktop: Release 10. Redlands, CA. Environmental Systems Research Institute.
European Commission, 2013. Overview of common agricultural policy (CAP) reform 2014–2020. Agricultural policy perspectives brief N°5.
FAO, 2010. Global forest resources assessment 2010 – main report. FAO forestry paper 163. Food and Agriculture Organization of the United Nations, Rome, Italy.
FAO, 2011. Assessing forest degradation. Towards the development of globally applicable guidelines. Forest Resources Assessment. Working Paper 177. Food and Agriculture Organization of the United Nations, Rome, Italy.
Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., 2005. Global consequences of land use. Science 309, 570–574.
Foody, G.M., 2002. Status of land cover classification accuracy assessment. Remote Sens. Environ. 80, 185–201.
Franklin, S., 2001. Remote Sensing for Sustainable Forest Management. Lewis Publishers, USA.
González, P., Lara, A., Gayoso, J., Neira, E., Romero, P., Sotomayor, L., 2005. Comparison of three methods to project future baseline carbon emissions in temperate rainforest, Curiñanco, Chile. Tropical Report. U.S. Department of Energy, National Energy Technology Laboratory, Morgantown, WV, USA.
González, Y., 2004. Óxidos de identidad: Memoria y juventud rural en el Sur de Chile (1935–2003). Tesis de doctorado en Antropología Social y Cultural. Tomo II, Anexos. Universitat Autònoma de Barcelona, Departament d'Antropologia Social i Prehistoria, Divisió d'Antropologia Social i Cultural, 224p.
Hantson, S., Chuvieco, E., 2011. Evaluation of different topographic correction methods for Landsat imagery. Int. J. Appl. Earth Observ. Geoinf. 13, 691–700.
Hartley, M.J., 2002. Rationale and methods for conserving biodiversity in plantation forests. Forest Ecol. Manage. 155, 81–95.
INFOR, 1986. Estadísticas forestales 1985. Instituto Forestal, División Estudios Económicos. Corporación de Fomento de la Producción. Santiago, Chile.
INFOR, 2010. Anuario Forestal 2010. Boletín Estadístico N° 128. Instituto Forestal, Ministerio de Agricultura. Santiago, Chile.
INFOR, 2013. Anuario Forestal 2013. Boletín Estadístico N° 140. Instituto Forestal, Ministerio de Agricultura. Santiago, Chile.
Iverson, L., Echeverría, C., Nahuelhual, L., Luque, S., 2014. Ecosystem services in changing landscapes: an introduction. Landscape Ecol. 29, 181–186.
IREN-CORFO, 1964. Informaciones meteorológicas y climáticas para la determinación de la capacidad de uso de la tierra. Santiago, Chile.
Klein Goldewijk, K., Beusen, A., de Vos, M., van Drecht, G., 2011. The HYDE 3.1 spatially explicit database of human induced land use change over the past 12,000 years. Global Ecol. Biogeogr. 20, 73–86.
Kole, C., Joshi, C.P., Shonnard, D.R. (Eds.), 2012. Handbook of Bioenergy Crop Plants. CRC Press, Boca Raton, Florida, USA.
Kröger, M., 2013. Global tree plantation expansion: a review. ICAS review paper series No. 3.
Kumar, R., Nandy, S., Agarwal, R., Kushwaha, S.P.S., 2014. Forest cover dynamics analysis and prediction modelling using logistic regression model. Ecol. Indicators 45, 444–455.
Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proc. Natl. Acad. Sci. 108, 3465–3472.
Lara, A., Little, C., Urrutia, R., McPhee, J., Álvarez-Garretón, C., Oyarzún, C., Soto, D., Donoso, P., Nahuelhual, L., Pino, M., Arismendi, I., 2009. Assessment of ecosystem services as an opportunity for the conservation and management of native forests in Chile. Forest Ecol. Manage. 258, 415–424.
Lara, A., Little, C., Nahuelhual, L., Urrutia, R., Díaz, I., 2011. Lessons, challenges and policy recommendations for the management, conservation and restoration of native forests in Chile. In: Figueroa, E. (Ed.), Biodiversity Conservation in the Americas: Lessons and Policy Recommendations. Santiago, Chile, pp. 259–299.
Laurance, W.F., Nascimento, H.E.M., Laurance, S.G., Andrade, A., Ribeiro, J.E.L.S., Giraldo, J.P., Lovejoy, T.E., Condit, R., Chave, J., Harms, K.E., D'Angelo, S., 2006. Rapid decay of tree-community composition in Amazonian forest fragments. Proc. Natl. Acad. Sci. 103, 19010–19014.
Li, H., Wu, J., 2004. Use and misuse of landscape indices. Landscape Ecol. 19, 389–399.
Lindenmayer, D.B., 2009. Large scale landscape experiments: Lessons from Tumut. CUP, Cambridge.
Liu, W., Song, C., Schroeder, T.A., Cohen, W.B., 2008. Predicting forest successional stages using multitemporal Landsat imagery with forest inventory and analysis data. Int. J. Remote Sens. 29, 3855–3872.
Lu, D., Mausel, P., Brondízio, E., Moran, E., 2003. Classification of successional forest stages in the Brazilian Amazon basin. Forest Ecol. Manage. 181, 301–312.
McGarigal, K., Cushman, S.A., Ene, E., 2012. FRAGSTATS v4: spatial pattern analysis program for categorical and continuous maps. Computer software program produced by the authors at the University of Massachusetts, Amherst.
Moran, E.F., Brondízio, E.S., Tucker, J.M., da Silva-Forsberg, M.C., Falesi, I., McCracken, S.D., 2000. Strategies for Amazonian forest restoration: evidence for afforestation in five regions of the Brazilian Amazon. In: Hall, A. (Ed.), Amazonia at the Crossroads: the Challenge of Sustainable Development. Institute for Latin American Studies, University of London, pp. 129–149.
Morrison, E.B., Lindell, C.A., 2010. Active or passive forest restoration? Restoration alternatives with avian foraging behavior. Restoration Ecol. 19, 170–177.
Murcia, C., 1995. Edge effects in fragmented forests: implications for conservation. Trends Ecol. Evol. 10, 58–62.
Myers, N., Mittermeler, R.A., Mittermeler, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.
Nahuelhual, L., Carmona, A., Lara, A., Echeverría, C., González, M., 2012. Land-cover change to forest plantations: proximate causes and implications for the landscape in south-central Chile. Landscape Urban Planning 107, 12–20.
Newton, A.C., Hill, R.A., Echeverría, C., Rey-Benayas, J.M., Golicher, D.J., Cayuela, L., Hinsley, S., 2009. Remote sensing and the future of landscape ecology. Prog. Phys. Geogr. 33, 528–546.
PCI, 2001. Using PCI Software. Richmond, Ontario.
Pedroni, L., 2003. Improved classification of Landsat Thematic Mapper data using modified prior probabilities in large and complex landscapes. Int. J. Remote Sens. 24, 91–113.
Pérez, C., 1999. Los procesos de descomposición de la materia orgánica de bosques templados costeros: interacción entre suelo, clima y vegetación. In: Armesto, J.J., Villagrán, C., Arroyo, M.T.K. (Eds.), Ecología de Los Bosques Nativos de Chile. Santiago, Chile, pp. 301–315.
Prach, K., Marrs, R., Pysek, P., van Diggelen, R., 2007. Manipulation of succession. In: Walker, L.R., Walker, J., Hobbs, R.J., (Eds.). Linking Restoration and Ecological Succession, New York, pp. 121–149.
Pütz, S., Groeneveld, J., Alves, L.F., Metzger, J.P., Huth, A., 2011. Fragmentation drives tropical forest fragments to early successional states: a modelling study for Brazilian Atlantic forests. Ecol. Modell. 222, 1986–1997.
Puyravaud, J.P., 2003. Standardizing the calculation of the annual rate of deforestation. Forest Ecol. Manage. 177, 593–596.
Reyes, R., Nelson, H., 2014. A tale of two forests: why forests and forest conflicts are both growing in Chile. Int. Forest. Rev. 16 (4).
Richards, J.A., Jia, X., 2006. Remote Sensing Digital Image Analysis. An Introduction, fourth ed. Springer, Germany.
Schnell, M.R., Pik, A.J., Dangerfield, J.M., 2003. Ant community succession within eucalypt plantations on used pasture and implications for taxonomic sufficiency in biomonitoring. Austral Ecol. 28, 553–565.
Schulz, J.J., Cayuela, L., Echeverría, C., Salas, J., Rey Benayas, J.M., 2010. Monitoring land cover change of the dryland forest landscape of Central Chile (1975–2008). Appl. Geogr. 30, 436–447.
Smith-Ramírez, C., 2004. The Chilean coastal range: a vanishing center of biodiversity and endemism in south american temperate rain forests. Biodiversity Conserv. 13, 373–393.
Smith-Ramírez, C., Díaz, I., Pliscoff, P., Valdovinos, C., Méndez, M.A., Larraín, J., Samaniego, H., 2007. Distribution patterns of flora and fauna in southern Chilean Coastal rain forests: integrating natural history and GIS. Biodiversity Conserv. 16, 2627–2648.
Song, X., Peng, Ch., Zhou, G., Jiang, H., Wang, W., 2014. Chinese grain for green program led to highly increased soil organic carbon levels: a meta-analysis. Nature 4, 4460.
TNC, 2011. Análisis multitemporal de imágenes en Cordillera Pelada: cuantificando el bosque nativo pre y post creación de la Reserva Costera Valdiviana. Valdivia, Chile.
Toro, J., Gessel, S.P., 1999. Radiata pine plantations in Chile. New Forest 18, 33–34.
Turner, B.L., Moss, R.H., Skole, D.L., 1993. Relating land use and global land-cover change: a proposal for an IGBP–HDP core project. Report from the IGBP–HDP working group on land-use/land-cover change. Joint publication of the International Geosphere-Biosphere Programme (Report No. 24) and the Human Dimensions of Global Environmental Change Programme (Report No. 5). Stockholm, Royal Swedish Academy of Sciences.
Vita, A., 1977. Crecimiento de algunas especies forestales nativas y exóticas en el Arboretum del Centro Experimental Forestal Frutillar, X Región. Boletín Técnico 47. Depto. Silvicultura, Universidad de Chile.
Vitousek, P.M., 1994. Beyond global warming: ecology and global change. Ecology 75, 1861–1876.
Vogelmann, J.E., Xian, G., Homer, C., Tolk, B., 2012. Monitoring gradual ecosystem change using Landsat time series analyses: case studies in selected forest and rangeland ecosystems. Remote Sens. Environ. 122, 92–105.

Williams, M.C., Wardle, G.M., 2005. The invasion of two native eucalypt forests by *Pinus radiata* in the Blue Mountains, New South Wales, Australia. Biol. Conserv. 125, 55–64.

Willson, M., De Santo, T.I., Sabag, C., Armesto, J.J., 1994. Avian communities of fragmented south-temperate rainforests in Chile. Conserv. Biol. 8, 508–520.

Zamorano-Elgueta, C., Cayuela, L., González-Espinosa, M., Lara, A., Parra-Vázquez, M.R., 2012. Impacts of cattle on the South American temperate forests: challenges for the conservation of the endangered monkey puzzle tree (*Araucaria araucana*) in Chile. Biol. Conserv. 152, 110–118.

Zamorano-Elgueta, C., Cayuela, L., Rey Benayas, J.M., Donoso, P.J., Geneletti, D., Hobbs, R.J., 2014. The differential influences of human-induced disturbances on tree regeneration community: a landscape aproach. Ecosphere 5 (7), art90.