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ORIGINAL ARTICLE

Native forest loss in the Chilean biodiversity hotspot: revealing the evidence

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Abstract The understanding of the spatial and temporal patterns in land use and land cover (LULC) change is a key issue for conservation efforts. In the Chilean hotspot, different studies have attempted to understand variations of LULC change. Nevertheless, a broader understanding of common patterns and variability of LULC over the entire range of the hotspot is lacking. We performed a complete review of the different studies reporting LULC changes and performed a joint analysis of their results using an integrated comprehensive approach. We related the variation of LULC change to latitude, time period and vascular plant richness using generalized linear models. Overall, there were nine studies, which covered 36.5 % of the study area, and reported the loss of 19 % of native forest (782,120 ha) between 1973 and 2011. The highest net forest loss was observed in the 1970–1990 period. This decreased in the 1990–2000 period and rose again in the 2000–2010 period. This result reveals a continuous forest loss in the last 40 years. Conversion of native forest to shrublands is the most important contributor to net native forest loss, accounting for 45 % of the loss. However, in the area of greatest species richness native forests are mainly converted to exotic tree plantations. Chilean forestry model has proved successful in expanding exotic tree plantation, but so far it has not been compatible with native forest conservation and restoration. It is imperative to design a new forestry policy to assure the conservation of one of the most unique biodiversity hotspots worldwide.

Keywords Land use and land cover change · Remote sensing · Temperate forest · Deforestation · Exotic tree plantation

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Introduction

The greatest current and future threat to biodiversity conservation is land use and land cover (LULC) change (Sala et al. 2000; Pereira et al. 2010), the effects of which may be faster and more obvious than those caused by climate change (Pereira et al. 2010). Forest loss, fragmentation and degradation is one of the most important causes of LULC change, with a clear negative effect on biodiversity and the provision of ecosystem services (Millennium Ecosystem Assessment 2005). Understanding how LULC change modifies wildlife habitat and ecosystems is a key issue for land planning oriented towards biodiversity conservation and the provision of ecosystem services.

Current and past LULC changes are the result of several political, economic, social, institutional, demographic and ecological factors (Geist and Lambin 2002). These factors evolve throughout time and configure multidimensional spatial and temporal scenarios operating as underlying forces driving LULC changes. They in turn generate different proximate causes of change. Timber production and extraction and the expansion of infrastructure and farming are the main drivers of forest loss. However, the intensity or extent of loss in each country or region is also determined by other specific biophysical factors in the landscape, such as soil quality, topography, landscape features, fires, drought and pests (Geist and Lambin 2002).

In Latin America, recent studies show great variations in the patterns of LULC change at regional and local level in different territorial units (Rodríguez et al. 2012; Armenteras et al. 2013; Kim et al. 2015). For example, Kim et al. (2015) report great variability in forest loss patterns among tropical Latin American countries. Countries with similar ecosystems (e.g. tropical humid) may present completely different change rates in forest cover loss (e.g. net annual change rate for 2005–2010: Brazil = −2.44 % vs. Guyana = 0.01 %). Change in patterns may vary widely within each country and even within each region of the country. In Colombia for example, the deforestation rate varies considerably between regions, differing by a factor of four or more (Armenteras et al. 2013). Rural population density, livestock grazing, subsistence and commercial agriculture, urban expansion and timber logging have been typically reported as important proximate causes explaining native forest loss within a single area (Hosonuma et al. 2012). However, significant differences can be also found at the local scale (e.g. Armenteras et al. 2013), resulting mostly from differences in the pattern of human occupation (Rodríguez et al. 2012). Thus, an understanding of spatial and temporal variations in LULC change is critical for conservation and management planning to be included in public policies and decision making.

There is a positive link between biodiversity and ecosystem services such as water provision, flood and debris flow occurrence, reduction of CO_2, pest regulation, pollination and recreation, species richness for timber production and/or freshwater fishing (Millennium Ecosystem Assessment 2005; Harrison et al. 2014). Highly diverse areas can be therefore especially vulnerable to LULC changes, particularly if such changes decrease the biodiversity. Currently, the 35 hotspots identified are inhabited by 2.08 billion people (31.8 % of human population) in just 2.3 % of Earth's land surface, but they support more than half of the world's plant species as endemics (Mittermeier et al. 2011; International Conservation 2014). Therefore, areas with high level of biodiversity and continuous threats (Myers et al. 2000; Mittermeier et al. 2004) need an international and coordinated effort for conservation planning. An understanding of how LULC changes proceed in these areas will provide critical insights in the potential threats of such areas and will therefore help to prioritize conservation efforts globally and nationally.

In Chile, native forest covers approximately 13.4 million hectares in different ecosystems. Chilean temperate forests represent more than half of the temperate forest in the Southern Hemisphere (Donoso 1993; CONAF 2011). Ecosystems located in central and southern Chile are considered to be one of 35 world biodiversity hotspots, due to their exceptional combination of a high concentration of endemic species and high level of threat (Myers et al. 2000; Mittermeier et al. 2004). Despite this, only 11 % of this global relevant area for biodiversity conservation is under the national protection system (International Conservation 2014). Additionally, this area has showed a rapid change in LULC (Echeverría et al. 2006; Schulz et al. 2010), and only a small part of conservation efforts have been directed towards the most threatened and diverse ecosystems (Armesto et al. 1998; Pliscoff and Fuentes-Castillo 2011; Squeo et al. 2012).

Different studies have attempted to understand the spatial and temporal patterns of LULC change and their proximate causes in areas included within the Chilean hotspot (e.g. Echeverría et al. 2006; Altamirano and Lara 2010; Zamorano-Elgueta et al. 2015). These studies provided different—and sometimes even contrasting—results in terms of rates of forest loss, dominant shifts towards other LULC classes, as well as the proximate causes of deforestation. For instance, Miranda et al. (2015) found an increasing forest loss through the time in one of the physiographical zones of Araucanía region (38.5° S). On the other hand, in a relatively close area (40°S) Zamorano-Elgueta et al. (2015) reports minimum native forest loss in the same period of time. Thus, it is key to analyse these patterns jointly across the entire Chilean hotspot in order to obtain a broader understanding of the spatial and temporal patterns and processes of LULC change in this region. International efforts, such as the REDD + program ("Reducing Emissions from Deforestation and Forest Degradation in Developing Countries") and other global initiatives to deal with global change effects (e.g. COP 21/CMP 11), reveal that the understanding of regional LUCC patterns is relevant for the design of public policies and conservation efforts. This is especially relevant in global biodiversity hotspots (Myers et al. 2000; Mittermeier et al. 2011) and is what the present study is intended for.

The main goal of the present study is to analyse native forest change over the last decades in the Chilean hotspot. This will help us to understand spatial and temporal forest change patterns over a large scale and identify the proximate causes that drive them at regional and national levels. The specific aims of this paper are: (a) to review and summarize the scientific evidence on native forest cover change in the Chilean hotspot; and (b) to analyse the spatial and temporal patterns of native forest cover change from 1970 to 2010.

Methods

Study area

The current delimitation of the Chilean biodiversity hotspot, also called "*Chilean winter rainfall–Valdivian forests*", extends from the Pacific coast to the Andean range and from the Antofagasta administrative Region to approximately General Carrera Lake in the Aisén Region (25°–47°S) (Arroyo et al. 2004). The hotspot includes various types of terrestrial ecosystems, different plant formations, wetlands, deserts and insular ecosystems. The main forests formations are distributed from the Valparaíso Region (33°S) southwards and include sclerophyllous forest, deciduous forest dominated by different species of the genus *Nothofagus* and the Valdivian and North Patagonian forests (Donoso 1993). We concentrated our research between the Valparaíso (33°S) and Los Lagos (42°S) Regions which contain the zones with the greatest vascular plant richness and endemism in Chile (Bannister et al. 2012) and the most populated areas. According to the national map of native forest cover in Chile (CONAF 2011), these area covers about 28 % of the country, containing 45 % of its native forest. It is also the area that suffers the greatest land use pressure in the country, due to the high concentration of some of Chile's principal economic activities. It contains 79 % of the country's urban and industrial zones, 94 % of its agriculture and 98.7 % of the exotic tree plantation (mostly *Pinus radiata* and *Eucalyptus* spp.) (CONAF 2011).

Spatial and temporal patterns of native forest cover change

We searched for scientific articles related to LULC change and native forest loss and fragmentation in the main databases, namely "ISI Web of Knowledge" and "SciELO". We used the keywords "Chile" + "deforestation" and "Chile" + "land use change" for the search.

We reviewed all articles found which dealt with LULC change in the sub-area of the Chilean hotspot (33°–42°S). The following selection criteria were applied: (i) the methodology used must include classification of satellite images taken at different times; and (ii) the classification process must be validated by ground control points. A table was constructed to summarize the analyses carried out in each study area, initially using four 11-year periods (1970–1980, 1980–1990, 1990–2000, 2000–2010) in which the studies were found. Because no studies reported results restricted exclusively to the decade 1980–1990, we decided to consider a single period to include the results of all studies conducted between 1970 and 1990. Therefore, our study analyses three different periods: 1970–1990, 1990–2000 and 2000–2010.

In order to assess the spatial and temporal variation of native forest cover, these changes were related to the period in which they occurred. We assigned the results reported in each study to one, two or three of the periods analysed, namely 1970–1990, 1990–2000 and 2000–2010. Each study was considered to belong to one of these three periods when at least 80 % of the duration of the study was included within that period. For example, if a study analysed LULC change in 1982–1992 and 1992–2001, the results were assigned to the 1970–1990 and 1990–2000 periods, respectively. We also obtained the following statistics that indicate the change in native forest cover for each period: (a) change in native forest (%): 100 − (final area * 100/initial area); (b) net change in native forest (ha): (final area − initial area); (c) net annual change in native forest (ha/year): (final area − initial area)/(final year − initial year); and (d) rate of change (%/year): Ln(final area/initial area) * (100/(final year − initial year) (Puyravaud 2003).

In those cases in which native forest loss was detected, the decrease in area was assigned to the main LULC classes to which forest cover was converted, namely: (e) agriculture and pasture land (ha); (f) exotic tree plantation (ha); and (g) shrubland (ha).

To calculate (e), (f) and (g), we calculated the area initially covered by native forest which was replaced by agriculture and pasture land, exotic tree plantation or shrublands. Note that the definitions of shrubland and native forest may vary between the different studies. For the purpose of this analysis, native forest includes old-growth and secondary forest. Native forest loss was considered as any change of a unit of area originally classified as native forest to another LULC. Shrublands include the sub-categories of dense shrublands and open shrublands, composed mainly of woody species of low height (native and exotic); it also includes arborescent shrublands, an intermediate state between shrublands and secondary forests.

In order to assess the relationship between forest loss (response variable) and spatial (i.e. latitude) and temporal (i.e. the period in which the change occurs) variables, forest cover change was characterized in four different ways: (i) deforestation rate (%/year); (ii) proportion of native forest replaced by agriculture and pasture land (%); (iii) proportion of native forest replaced by exotic tree plantation (%); and (iv) proportion of native forest in areas subsequently covered by shrublands (%). Two additional covariates might contribute to explain patterns of forest loss: stage of landscape transformation, quantified as the initial native forest cover as percentage of each study area in each period (%), and native plants species richness as reported by Bannister et al. (2012). However, the correlations between latitude and percentage of the study area covered by native forest in the initial year was high ($r = 0.785$), implying that southwards the study region the percentage of native forest increases. We therefore decided to exclude the latter from the analyses in order to avoid multicollinearity. The relationship between the four different responses (i, ii, iii, iv) and the explanatory variables (latitude, time period and native plant species richness) was analysed by means of generalized linear models, using a Gaussian error distribution. Candidate models were compared using the Akaike information criterion corrected for small sample sizes ($\mathrm{AIC_c}$, Burnham and Anderson 2002), where the most complex model was:

$$Y_{it} \sim \mathrm{Period}_t \times \mathrm{Latitude}_i + \mathrm{Richness}_i + \varepsilon_i,\ \ \varepsilon_i \sim N(0,\ \sigma^2)$$

where Y_{ij} refers to the value obtained from each of the i study areas at period t for each response variable, Period_t was a factor with $t = 3$ levels (1970–1990, 1990–2000, 2000–2010), $\mathrm{Latitude}_i$ was a covariate, whose effect on the response variable might change depending on the period t (i.e. interaction term between Period and Latitude), and $\mathrm{Richness}_i$ was a covariate representing native tree species richness in each of the i study areas. ε_i responds to the error term of observation i, which follows a Gaussian distribution. We formerly attempted to model the response variables by means of generalized linear mixed models with a random error associated with the study site, but comparison of models with and without random effects using the $\mathrm{AIC_c}$ indicated a negligible effect of the random term. Therefore, for the sake of simplicity, we showed the results of generalized linear models excluding the random error term associated with the study site.

Models with a difference in $\mathrm{AIC_c} < 2$ units are considered to have equivalent empirical support, whereas a difference value within only 4–7 units of the best model has considerably less support. Differences in $\mathrm{AIC_c} > 10$ indicate that the worse model has virtually no support and can be omitted from further consideration. Deviance was used as a measure of discrepancy to assess the model's goodness of fit. Deviance reduction or explained deviance (D^2) is estimated as:

$$D^2 = (\text{null deviance} - \text{residual deviance})/\text{null deviance}$$

where the null deviance shows how well the response variable is predicted by a model that includes only the intercept. All analyses were performed using the R environment (R Core Team 2014).

Results

Nine studies coincided with our selection criteria: Echeverría et al. (2006), Aguayo et al. (2009), Altamirano and Lara (2010), Schulz et al. (2010), Echeverría et al. (2012), Altamirano et al. (2013), Vergara et al. (2013), Miranda et al. (2015) and Zamorano-Elgueta et al. (2015). These studies reported results from 13 different study areas (Fig. 1). All the studies are based on medium spatial resolution satellite imagery such as Landsat and Aster, with a supervised maximum likelihood method to classify LULC classes. The total area analysed in these studies represents 36.5 % of the national territory between Valparaíso (33°S) and Los Lagos Region (42°S). Only one of seven administrative regions within the study area, Libertador Bernardo O'Higgins Region (34–35°S), was not covered by any study.

Only two studies (Schulz et al. 2010; Miranda et al. 2015) quantified native forest loss in all three periods (1970–1990, 1990–2000, 2000–2010); five studies covered only two of the periods (Echeverría et al. 2006, 2012; Altamirano et al. 2013; Vergara et al. 2013; Zamorano-Elgueta et al. 2015), and one study (Altamirano and Lara 2010) analysed only one period. The study of Aguayo et al. (2009) is an exception because it analyses changes in land cover over a prolonged period (1979–2000), extending to two of the periods considered in the present work. The results of this study were therefore included in the summary table for descriptive purposes (Table 1), but could not be assigned to any of the study periods defined and were therefore not included in the analyses.

In 12 of 13 study areas net losses of native forest are reported for all periods. The total net native forest loss reported for all areas during the whole period considered by this study was 782,120 ha, equivalent to 19 % of the total native forest cover at the beginning of the period in

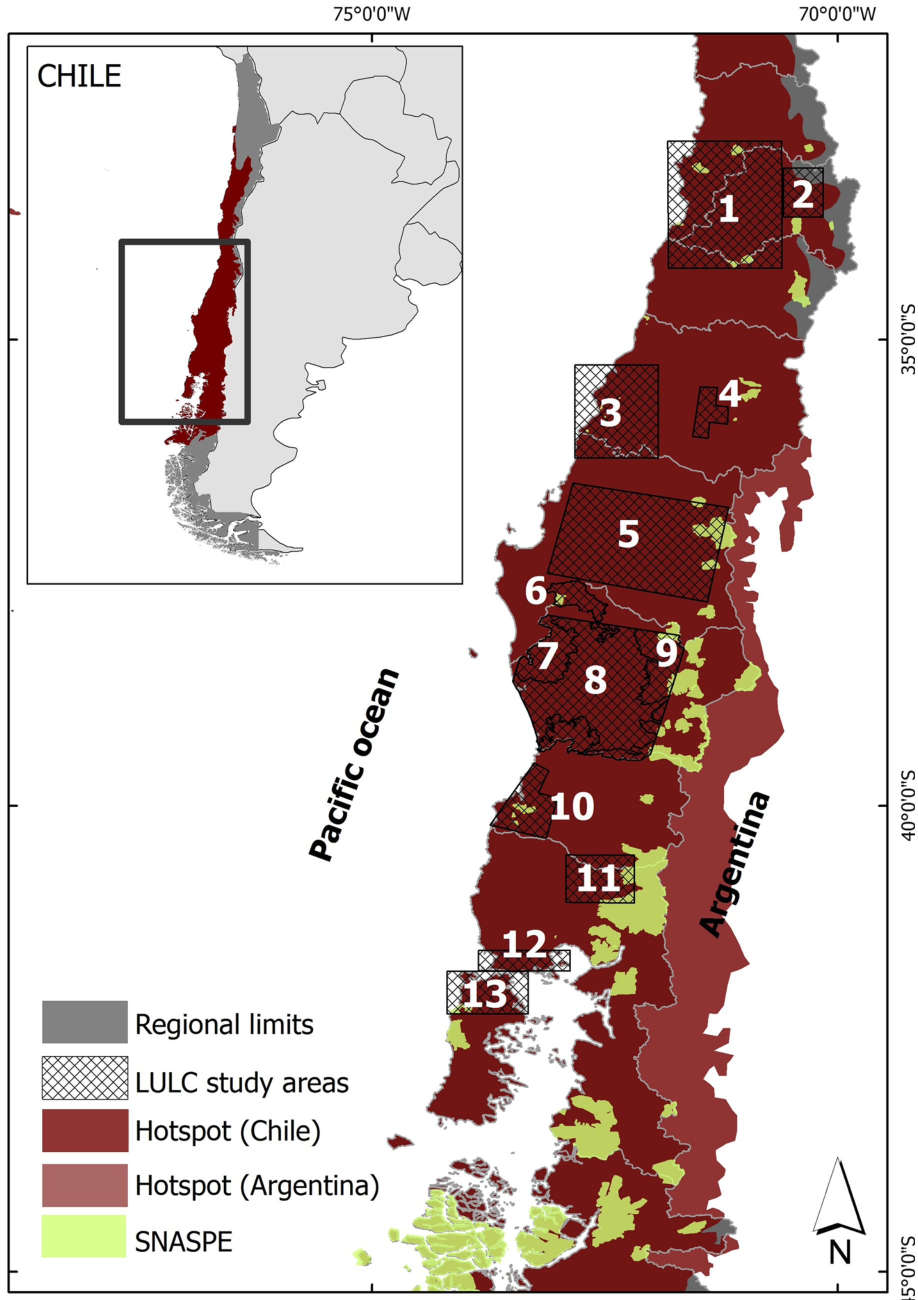

Fig. 1 Geographical distribution of LULC change studies in the Chilean hotspot. *1* Schulz et al. (2010), *2* Vergara et al. (2013), *3* Echeverría et al. (2006), *4* Altamirano and Lara (2010), *5* Aguayo et al. (2009), *6* Altamirano et al. (2013), *7–9* Miranda et al. (2015), *10* Zamorano-Elgueta et al. (2015), *11–13* Echeverría et al. (2012). The study areas boundaries are for reference only

each study area. Overall, a greater mean annual net native forest loss was observed in the first period (1970–1990) than more recent periods, diminishing by 77 and 51 % for 1990–2000 and 2000–2010 period, respectively (Table 1: ANFL). There was also a general increase in native forest loss in the period 2000–2010 as compared to 1990–2000. This pattern of native forest loss is also reflected by the native forest loss rate (NFLR), which is lower in 1990–2000 (mean NFLR: 1.6 %/year) than in the previous period (mean NFLR: 2.9 %/year); however, it rises again in 2000–2010 (mean NFLR: 2.4 %/year). This pattern is also reflected in a higher forest loss (net and percentage) in 2000–2010 than in 1990–2000 for the most of the geographical zones (Fig. 2).

Spatial and temporal patterns of native forest cover change

In the northern part of the study area (33°S), we observed that the predominant land cover type is shrublands. From the Maule Region (35°S) to the Los Ríos Region (40°S), the landscape is dominated by productive activities, with tree plantation of exotic species being the main land use to which native forest changed. The land uses to which native

Table 1 Summary of studies of LULC change

Period	Study area	NFL (%)	ANFL (ha)	NFLR (%/year)	AL (ha)	TPL (ha)	SL (ha)
1970–1990	5	28	8795	1.6	10,110	132,388	40,138
	1	31	6324	3.7	2530	0	50,608
	7	21	2585	1.7	17,246	13,043	6277
	8	38	4233	3.4	52,119	2813	5818
	9	10	1521	0.7	15,793	830	6612
	3	53	4257	5.1	4800	34,798	43,197
Mean (CV)		31 (0.5)	3784 (0.5)	2.9 (0.6)	18,498 (1)	10,297 (1.4)	22,502 (1)
Total period					92,488	51,484	112,512
1990–2000	6	7	262	0.5	4499	5282	993
	13	2	236	0.1	2303	0	16,694
	11	10	1206	0.9	0	0	25,905
	12	10	1077	0.8	1836	0	18,719
	4	44	1609	4.1	311	13,974	14,723
	7	23	2626	2.2	10,966	17,932	4535
	8	22	1760	2.1	15,002	4094	3784
	9	12	2061	1.1	4111	6284	17,709
	1	4	376	0.3	2530	0	2530
	3	31	1713	3.6	2730	19,501	2730
	2	31	1571	3.8	–	–	–
	10	5	637	0	0	19,980	8910
Mean (CV)		17 (0.8)	1261 (0.6)	1.6 (0.9)	4026 (1.2)	7913 (1)	10,657 (0.8)
Total period					44,288	87,047	117,232
2000–2010	1	13	1903	1.5	5060	5060	9963
	13	13	2884	1.7	0	0	24,283
	11	8	1206	1	1568	0	12,001
	12	11	1884	1.5	0	0	25,755
	7	35	4184	4.8	7404	29,930	2873
	8	25	2081	3.2	13,847	6801	1255
	9	14	2727	1.7	4087	12,123	6507
	6	20	1032	2.5	4312	5433	618
	2	44	1506	5.8	–	–	–
	10	0	17	0.01	3240	17,550	3780
Mean (CV)		18 (0.7)	1942 (0.6)	2.4 (0.7)	4391 (1.0)	8544 (1.1)	9671 (1.0)
Total period					39,518	76,897	87,035
Total					186,404	347,816	356,917

Study area: 1—Schulz et al. (2010), 2—Vergara et al. (2013), 3—Echeverría et al. (2006), 4—Altamirano and Lara (2010), 5—Aguayo et al. (2009), 6—Altamirano et al. (2013), 7–9—Miranda et al. (2015), 10—Zamorano-Elgueta et al. (2015) and 11–13—Echeverría et al. (2012)

NFL native forest loss in the period (%), *ANFL* annual net native forest loss (ha), *NFLR* native forest loss rate for the period (%/year), *AL* native forest loss replaced by agriculture and pasture land (ha), *TPL* native forest loss replaced by exotic tree plantation (ha) and *SL* native forest loss replaced by shrublands (ha), – no information, *CV* coefficient of variation

forest is lost and the intensity of native forest loss vary as a function of the geographical zone in which the process occurs (Table 2; Fig. 3). For example, although clearing for agriculture and pasture land is extensive throughout the whole study area, it is not the main driver of landscape change. When all the study areas are considered for the whole period analysed (1973–2010), conversion of native forest to shrublands is found to be the main change, accounting for 45 % of native forest loss. This is true for all the periods covered by this study. However, this type of change is concentrated mostly in the northernmost and southernmost regions of the study area, while between the Maule Region (35.5°S) and the coastal area of the Araucanía and Los Ríos Regions (40°S) exotic tree plantation represents the main land use to which native forest has been converted. In this zone where deforestation for exotic

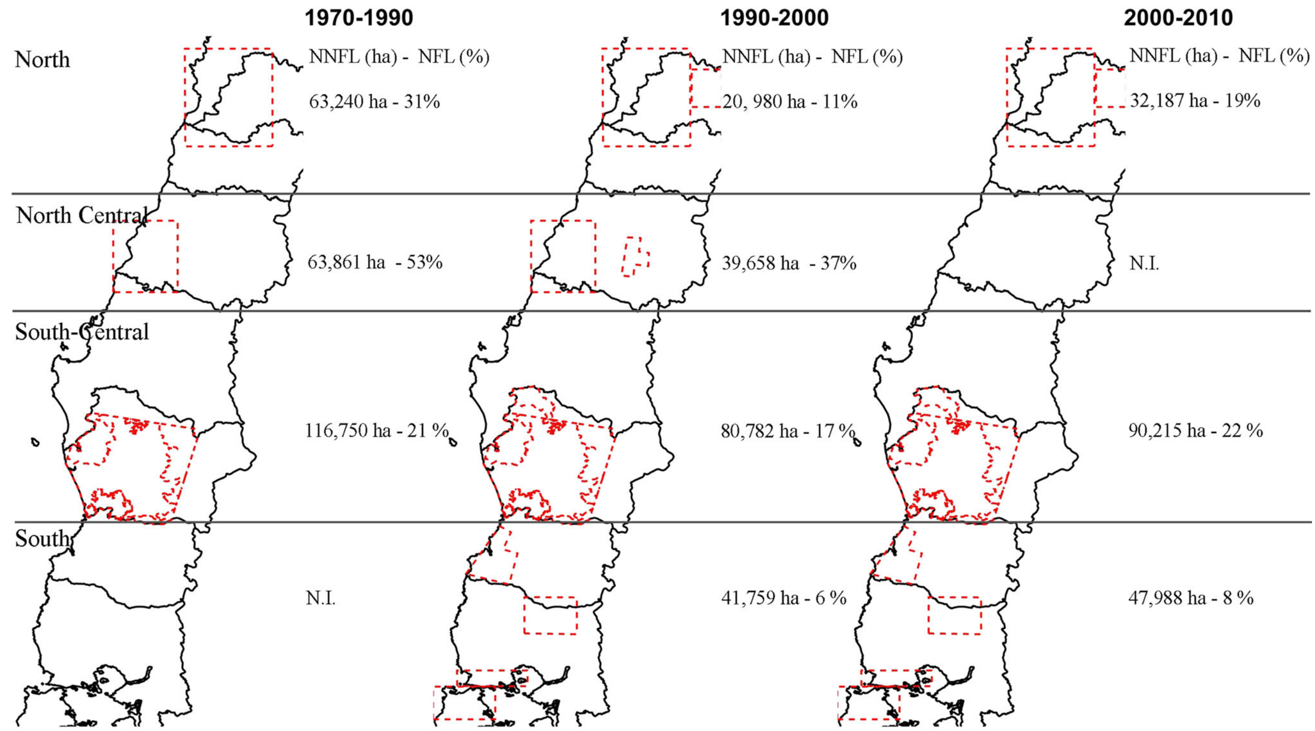

Fig. 2 Geographical pattern of native forest loss for different periods analysed. *Red dashed line* reference location of study areas. *Irregular solid black lines* administrative regions in the study area. *NNFL* net native forest loss (ha), *NFL* native forest loss in the period (%), *N.I.* no information (colour figure online)

Table 2 Generalized linear model of relationship between different response and predictors variables

Response variable	Predictor variables	Residual deviance	D^2	Coefficients	SE
Deforestation rate	Null model	58.2			
	Latitude	35.3	0.39	−0.332	0.095
Native forest changed to exotic tree plantation	Null model	14,842.4			
	Latitude	14,046.7	0.12	−4.478	1.682
	Richness	6573.2	0.56	0.465	0.109
Native forest changed to shrublands	Null model	23,371.7			
	Latitude	20,677.3	0.12	7.111	1.768
	Richness	7266.4	0.69	−0.623	0.115

D^2 = deviance reduction. Change to agriculture and pasture land is not showed because none of predictor variables improved the null model

tree plantation is highest, it is concentrated the greatest richness of native plant species in the hotspot (Fig. 3).

The best-fit model for native forest change only included the effect of latitude, which explained ca. 39 % of the variability (Table 2). Deforestation rate diminished southwards (Fig. 4a), where a larger proportion of the land was covered by native forest (correlation between latitude and percentage of the study area covered by native forest in the initial year was $r = 0.785$). Although in general a greater net annual native forest loss is observed between 1970 and 1990, the deforestation rate does not show any temporal trend during the whole period analysed (1970–2010) (Appendix 1 in electronic supplementary material). The land cover categories to which native forest changed were also more affected by geographical variation than by time period. The best-fit models for native forest replacement by exotic tree plantation and shrublands included both latitude and species richness, whereas native forest replacement by agriculture and pasture land was not related to any explanatory variable at all (Appendix 1 in electronic supplementary material). Latitude was negatively related to forest replacement by exotic tree plantations, whereas richness of native plant species was positively related (Table 2; Fig. 4). Native forest replacement by shrublands, on the other hand, was positively related to latitude and negatively related to species richness (Table 2; Fig. 4).

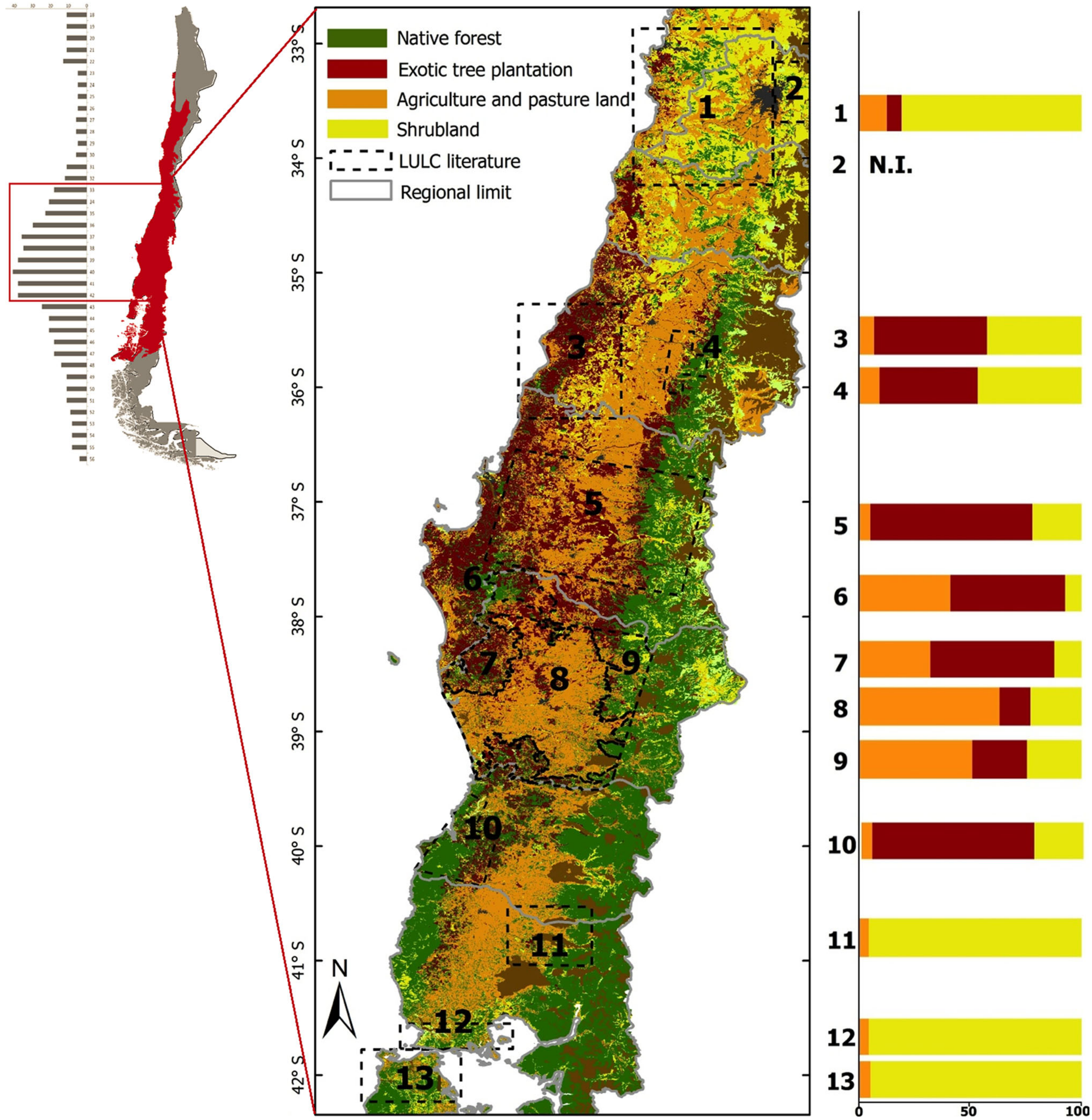

Fig. 3 Geographical distribution of the main land covers, richness of tree species, and percentage of change from native forest to other land covers. *Left panel* relative number of vascular plant species for each degree of latitude, based on the data from Bannister et al. (Bannister et al. 2012). *Centre panel* land covers types and location of study areas analysed by the different studies. *Right panel* relative importance (%) of the change from native forest to different categories of land cover shown in the same colours as in the central panel. *N.I.* no information. LULC information from Native Vegetation Survey of Chile (CONAF 2011)

Discussion

It is commonly recognized that major native forest loss in the Chilean hotspot (33–42°S) began with the Euro-Chilean colonization of the mid-nineteenth century (Donoso and Lara 1995). This change was concurrent with changes in other Latin American countries as a result of government-promoted colonization policies (Armenteras and Rodriguez 2014). At that time, the Chilean Government decided to incorporate the southern part of its national territory and effectively connect it to the centre of the country by promoting the settlement of regions between

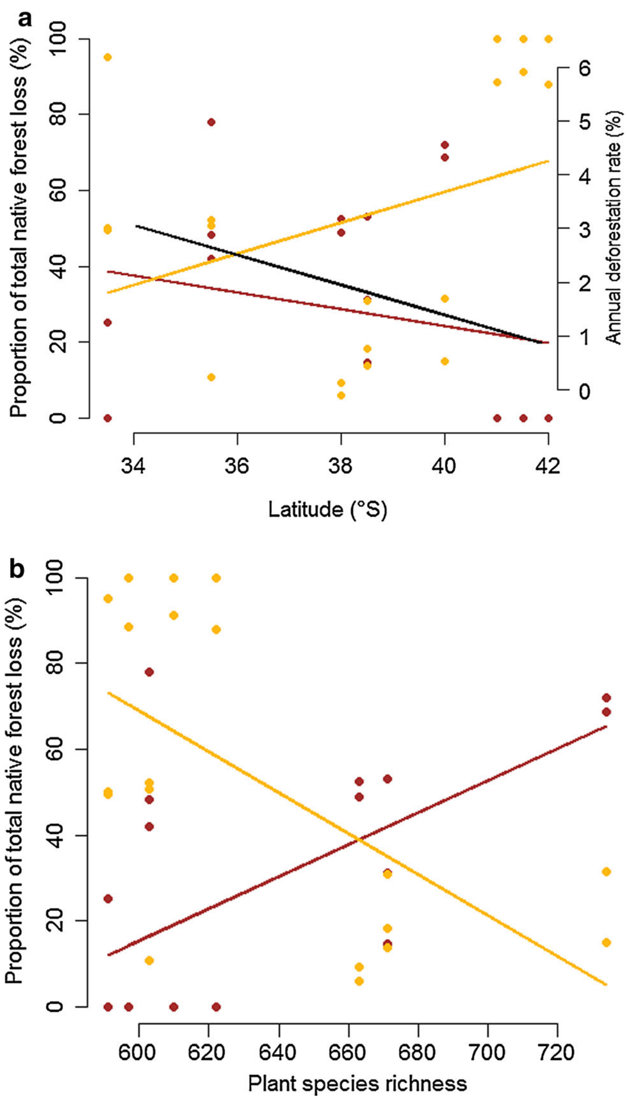

Fig. 4 Relationship between proportion of total native forest loss as a function to which LULC changed with **a** latitude, and **b** species richness. *Yellow line* predicted model of change to shrublands. *Yellow points* observed change to shrublands. *Brown line* predicted model of change to exotic tree plantation. *Brown points* observed change to exotic tree plantation. *Black line* annual deforestation rate (%). Change to agriculture and pasture is not showed because none of predictor variables improved the null model (colour figure online)

38° and 42°S. Large areas were burnt for conversion to arable and grazing land, especially in the Central Valley and the valleys and foothills of the Andes and the Coastal Range. Native forests were also cleared for the foundation of towns and the construction of the railway line to the south between 1880 and 1913 (Otero 2006; Armesto et al. 2010; Lara et al. 2012). This process was concentrated principally south of 37°S, since further north, where the main commercial centres and cities were located, pressure on the forests had already begun in an earlier period. All this process is associated with the conquest and the colonial period starting in 1550 (Lara et al. 2012) progressing from north to the south which is partially supported by the high correlation between the initial native forest cover (%) of each study area and latitude. During the 1900s, the Chilean hotspot continued to lose extensive areas of native forest due to catastrophic fires set intentionally to open up land for agriculture and cattle raising (Otero 2006; Armesto et al. 2010; Lara et al. 2012). Another, more recent, historical event recognized as one of the underlying driving forces of native forest loss was the establishment of Decree Law 701 in 1974, which remained in force until 2012. This decree was passed to provide an incentive for the afforestation of areas with degraded soils considered suitable for forestry. However, its practical effect was an economic incentive for the expansion of the forestry industry with plantations of exotic tree species, especially *Pinus radiata*, *Eucalyptus globulus* and *E. nitens* (Lara and Veblen 1993; Reyes and Nelson 2014). This resulted in a high annual increase in the total area planted with exotic species, which rose from 300,000 ha in 1970 to 3.047 million hectares in 2016 (Lara and Veblen 1993; CONAF 2016).

This rapid expansion of the forestry industry has led other countries to consider the Chilean forestry model as a productive model to be imitated (Lara and Veblen 1993), and the main Chilean forestry companies have expanded their business into countries such as Argentina, Brazil and Uruguay (Reyes and Nelson 2014). Consistent with this trend, FAO (2006) projects an increase in exotic tree plantation in Latin America and the Caribbean from 11.8 million hectares in 2003 to 17 million hectares in 2020, concentrated mainly in Brazil, Chile and other countries of South America. The biophysical potential of the region for exotic tree plantations is estimated at more than 70 million hectares (FAO 2006). In Chile, the negative impacts of this productive model have already become apparent, especially on water availability, social conflicts and biodiversity conservation (Lara and Veblen 1993; Little et al. 2009; Lara et al. 2011; Nahuelhual et al. 2012; Reyes and Nelson 2014).

According to the latest estimates of the state of Chilean forests in the FAO report (2010), Chile is in post-transition phase four, typical of countries with more than 50 % forest cover and a net forest gain (Hosonuma et al. 2012). It should be noted that this statement is based on a definition of "forest cover" which includes both native forest and plantations of exotic tree species (FAO 2010). The inclusion of such exotic tree plantations in the definition of "forest" has been questioned (Sasaki and Putz 2009; Putz and Redford 2010), mainly because of the detrimental effects that they have on biodiversity conservation (Putz and Romero 2014) and the provision of ecosystem services (Little et al. 2009; Lara et al. 2009, 2011; Nahuelhual et al.

2012), as well as the generation of social conflicts (Lara et al. 2012; Reyes and Nelson 2014). This indicator of total forest cover used by FAO (2010) does not provide information on the reduction of area and consequent loss of health and integrity of Chilean native forests, as well as the loss of biodiversity and the ecosystem services that these ecosystems provide. This adds evidence to the need to replace the criteria used by FAO for the assessment of changes in forest cover in different countries with one that separates changes in the area of native forests from changes in exotic tree plantations.

Patterns of native forest loss in the Chilean hotspot since the 1970s

We found a great variation in patterns of LULC change. For example, Schulz et al. (2010) and Echeverría et al. (2006) indicated that the net native forest loss occurred principally in the period 1970–1990, reporting figures of 74 and 78 % total forest loss, respectively. Miranda et al. (2015), in the Araucanía Region, observed different loss patterns as a function of the geographical area analysed. In area 8, Miranda et al. (2015) found a similar pattern to that observed by Schulz et al. (2010) and Echeverría et al. (2006), with 60 % of the total loss occurring in the first period (Fig. 1). However, in areas 7 and 9 the authors found a different pattern with a relatively constant loss in all three periods (Fig. 1). Echeverría et al. (2012), considering all three study areas, and Altamirano et al. (2013) found that there was a larger net native forest loss in the period 2000–2010, reporting increases of 46 and 272 %, respectively, over the previous period (1990–2000). Vergara et al. (2013) reported a very similar net native forest loss in these two periods. Nevertheless, in the most recent period they found the highest deforestation rate recorded in Chile (5.8 %). Zamorano-Elgueta et al. (2015) described a greater net native forest loss in the period 1990–2000 (not covering 1970–1990), with a net annual loss in the most recent period of only 17 ha. They attribute this to the high rate of natural regeneration, covering an area close to 23,000 ha.

The different studies considered in this analysis include 36.5 % of the area in Chile between the Valparaíso (33°S) and Los Lagos (42°S) region and may be considered representative of the LULC dynamics and forest loss associated with conversion to other land uses over this range. The studies cover the three major physiographical regions of Central and south-central Chile: Coastal Range, Central Valley and low and mid-elevations of the Andes Mountains. With the exception of Aguayo et al. (2009), the analyses do not include higher elevation slopes in the Andes, where the lowest intensity of forest loss is to be expected due to their inaccessibility and legal protection. Thus, these studies are distributed across the areas where there is the greatest pressure on land use, although they also include some state-protected areas.

Native forest loss is occurring in all the areas studied in all periods. The evidence from the various studies shows widespread, rapid destruction of native forests and conversion to other land uses since the 1970s. However, our results show that there is great variability in native forest loss, in terms of intensity and the proximate causes of loss. This may be expected as a response to the great variability of environmental and climatic conditions and the accessibility and economic development of the different administrative regions, as observed in other Latin American countries (Aguiar et al. 2007; Gasparri and Grau 2009; Armenteras et al. 2013; Armenteras and Rodriguez 2014).

The net loss has diminished in recent periods in comparison with 1970–1990. Possible causes for this decrease are that international markets have also begun to require certification of forestry companies mainly by the Forest Stewardship Council (FSC). This has resulted in better internal control, making the conversion of native forests to exotic tree plantation less feasible, and obviously there are less remnants of native forest. However, the increased frequency of forest fires in Chile (González et al. 2011), the continuous unregulated logging of forests (Donoso et al. 2014), the increase in the use of firewood from native species for domestic and industrial heating (Universidad de Chile 2008) suggest that forest loss due to conversion to other LULC as well as forest degradation will continue in the future.

Other variables that have an impact on deforestation are the environmental conditions in each area, since suitable conditions for the expansion of productive activities (e.g. crops, cattle ranching and exotic tree plantation) may help to increase land use pressure on native forest. For example, Miranda et al. (2015) confirm this general trend in which areas with more suitable environmental conditions present a lower proportion of native forest cover, while the proportion increases steadily towards the less accessible zones or those with adverse environmental conditions. This explanation is true at a national scale. The most heavily disturbed landscapes, in which native forests cover a small proportion of the total area and are reduced to small fragments in a matrix dominated by other land uses, are more susceptible to deforestation (Echeverría et al. 2007). In these landscapes, there is better access to the remnant forest fragments (e.g. roads, towns, cities, ports) and a greater demand for firewood and other forest products. Landscapes where productive use predominates tend to expand and coalesce become more homogeneous. For example, the presence of exotic tree plantations is spatially contagious (Altamirano et al. 2013), leading to the appearance of young plantations probably due to the reduced need for new investment in roads and transport.

Different phenomena are important for the future development of the landscape within the biodiversity hotspot in Chile. In the northern part, the landscape is dominated by shrublands, especially the so-called *Acacia caven* savannahs (Schulz et al. 2010). This vegetation is replacing native forests, particularly in more arid areas as the species is highly persistent (Van de Wouw et al. 2011). Another important aspect is the possible future evolution of exotic tree plantation in the territory. Today, the greatest concentration of exotic tree plantations is located between the Maule (35°S) and Araucanía Regions (39°S). This expansion began from the Maule and Bio–Bio Regions (37.5°S), where the greatest increase of exotic tree plantation started in 1974. One of the highest annual deforestation rates ever reported for Chile (5.1 %) was observed in the Coastal Range in the south of the Maule Region (Echeverría et al. 2006). Meanwhile, in the same period the deforestation rate for the Coastal Range in the Araucanía Region was 1.7 %, and in 1987 exotic tree plantation covered only 15 % of the area. This temporal pattern of the deforestation rate as a function of latitude suggests the southwards expansion of exotic tree plantation. Echeverría et al. (2012) partially confirm this southwards pattern in the spread of exotic forest plantations: in the northernmost area studied (40.8°S), exotic tree plantations first appear in the analysis in 1998 and increase in the following period, while in the other two study areas further south (41.5°S and 42.5°S) they first appear in 2007, occupying a proportion close to 1 % of each study area. This southwards shift of forest commercial plantations is probably a response to land use saturation, in which the majority of land available for forestry expansion and land owners willing to sell their land to the timber companies was exhausted early on, forcing the forestry industry to migrate to other regions, preferably further south where more suitable soils and climatic conditions exist, or even to Argentina, Uruguay and Brazil (FAO 2006). Our results also confirm this pattern with the exception of the study from Vergara et al. (2013), where the highest deforested areas in 1970–1990 then diminished in more recent period. Likewise, lowest deforested areas in the first period then increased the deforestation rate in recent periods. This pattern is particularly evident in southern areas where the deforestation rate increased in 2000–2010 compared to 1990–2000.

Conclusions

Our results show a generalized loss of native forest in the Chilean biodiversity hotspot which remains at recent times. The annual rate of forest loss ranges between 0 and 5.8 %, with an average of 2.1 %. The highest net forest loss mainly occurred between 1970 and 1990. The figure fell in 1990–2000 but rose again in the most recent period (2000–2010). The conversion of native forest to shrublands has been a permanent and common LULC change since the 1970s and the most significant for net native forest loss (accounting for 45 % of the loss). However, in the area of greatest tree species richness native forests are mainly converted to exotic forest plantations. The effects of the expansion of exotic tree plantations become apparent in the 1990s, when they start to dominate large areas of central Chile.

There are significant contrasts between the product types, market profitability and ecosystem services provided by exotic forest plantations and native forests. The certification schemes applied by large timber companies should therefore promote forest policies and decisions which will protect landscapes and watersheds by combining these two and other land uses. This would be the basis for a balanced provision of market goods (e.g. timber) and ecosystem services (e.g. water supply quantity and quality), maximizing the benefits to the society. To reach this balance, it is crucial that the government should increase significantly the incentives for sustainable management, conservation and restoration of native forests, with a special consideration for prevent loss and degradation of native forests. However, the Chilean Congress is discussing the renewal of the Law that has provided incentives to commercial exotic tree plantations between 1974 and 2012 (DL 701), considering payments totalling US$50 million for the period 2016–2023. This goes directly against these recommendations and would promote homogeneous landscapes dominated by exotic plantations, reducing the chances of the proposed balance between land uses for the simultaneous production of timber and ecosystem services. The Chilean forestry model has proved successful in expanding exotic tree plantation, but this productive model shows that so far it has not been compatibility with native forest conservation. It is imperative to design a new forestry policy to assure the conservation of one biodiversity hotspots in the globe.

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