

Impact of land-use change on biodiversity and ecosystem services in the Chilean temperate forests

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Abstract

Context Land-use change impacts biodiversity and ecosystem services, which are intrinsically related. There is a serious lack of knowledge concerning on how land-use change affects this relationship at landscape level, where the greatest impacts have been reported. A proper knowledge of that relationship would provide crucial information for planning conservation strategies. The forest landscape of southern Chile, which includes Valdivian Temperate Forest, has been designated as a hotspot for

biodiversity conservation. However, this landscape has been transformed by land-use change.

Objective We evaluated the impact of land-use change on the spatial patterns of the diversity of native forest habitat and the influence of these impacts on the provision of the ecosystem services water supply, erosion control, and organic matter accumulation from 1986 to 2011.

Methods The evaluation, at the landscape level, was carried out using satellite images, landscape metrics, spatially explicit models and generalized linear models. Results: We found that the area loss of native forest habitat was 12%, the number patches of native forest habitat increased more than 150% and the Shannon diversity index decreased by 0.20. The largest decrease in the provision of services was recorded for erosion control (346%), and the smallest for water supply (11%).

Conclusions The loss of provision of the ecosystem services can be explained by the interaction between the area loss, increase in the number patches and diversity loss. We recommend that the conservation planning strategies should consider the current landscape configuration, complemented with land-use planning.

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Keywords Anthropogenic processes · Changing landscape · Habitat diversity · Natural capital · Spatially explicit models

Introduction

In the last century, impactful changes in forest landscapes in the world have been reported (Wu 2013). These changes have been generated by global change drivers, resulting in the reduction of biodiversity, which is broadly defined as the variety and abundance of genes, species, and ecosystems (Butchart et al. 2010). As a result of these changes, the United Nations Organization (UNO) promulgated the Convention on Biological Diversity in 1992, which has as one of its objectives to achieve a significant reduction in the rate of biodiversity loss. In 2005, the Millennium Ecosystem Assessment (MA) issued two important declarations: (i) the intrinsic relationship between biodiversity and ecosystem services, which are broadly defined as the benefits provided by ecosystems that contribute to making human life both possible and worth living; and (ii) the conservation of biodiversity justifies the protection of the ecosystem services (Millennium Ecosystem Assessment 2005). These declarations have generated positive changes in the reduction of biodiversity loss and changes in the provision of the ecosystem services (Carpenter et al. 2009). However, there are few studies that promote a better understanding on how global change drivers impact biodiversity and how these changes influence the provision of ecosystem services (Schneiders et al. 2012; Iverson et al. 2014).

Land-use change and land-use intensification (e.g., agricultural and commercial plantations) has been identified as the main global change driver worldwide (Butchart et al. 2010). Land-use change and land-use intensification can lead to habitat loss and fragmentation, and these are two of the greatest threats to forest landscape (Lindenmayer 2012). Land-use change and land-use intensification modifies the spatial patterns (composition and spatial configuration) of forest landscape, such as total habitat area, habitat patch density and richness of habitat patches (Lindenmayer 2016). Consequently, these alterations can cause biodiversity loss at different levels (Lindenmayer 2012).

Biodiversity loss usually involves alterations in the provision of ecosystem services (Millennium Ecosystem Assessment 2005). These alterations highlight the need to understand the relationship between biodiversity and ecosystem services. Currently, this

complex relationship has not been widely studied (Chan et al. 2007; Costanza et al. 2007; Schneiders et al. 2012). Only a few studies reported that the increase in plant species richness positively affects the biomass and primary production (Hooper et al. 2002; Pfisterer and Schmid 2002; Díaz et al. 2006; Costanza et al. 2007; Hector and Bagchi 2007). Other studies reported that the species richness and the presence of endangered species are directly linked to the provision of ecosystem services, which are ecotourism and medicines (Swift et al. 2004; Hooper et al. 2005; Thomas et al. 2008; Elmqvist et al. 2010). Moreover, other studies have reported that a few dominant species and functional group of species are directly linked to the provision of timber, food and water purification (Hooper et al. 2002, 2005; Luck et al. 2009). Although these studies have generated a better understanding of the relationship between biodiversity and ecosystem services, additional studies should be performed to understand how different levels of biodiversity (e.g. populations, communities, habitat among others) are related to the provision of the ecosystem services (Costanza et al. 2007; Schneiders et al. 2012). The knowledge of this relationship will contribute to the planning conservation strategies to both resources (Iverson et al. 2014).

The greatest impacts on biodiversity have occurred at the native forest habitat level, such as the reduction and habitat loss (Hooftman and Bullock 2012; Liu et al. 2014; Ren et al. 2016), generating important changes in the provision of ecosystem services in different regions of the world (Wu 2013). The impacts on the diversity of native forest habitat and ecosystem services have occurred due to the increase of the human population (Vihervaara et al. 2010; Pan et al. 2013), which has transformed forest landscapes to anthropogenic landscapes in the last decades (Bai et al. 2011). The need to conserve habitat diversity and ecosystem services within forest and anthropogenic landscapes has been recognized by practitioners (Vihervaara et al. 2010; Iverson et al. 2014). The conservation of both resources would help to ensure the maintenance of multiple benefits for human populations that inhabit them (Vihervaara et al. 2010; Delgado et al. 2013).

The forest landscape of southern Chile, which includes Valdivian Temperate Forest, has been designated as a high priority area for biodiversity

conservation in the world (Myers et al. 2000). Several authors have registered that this forest landscape supports the provision of key ecosystem services, such as water supply, which are essential not only for human life but also for agricultural and aquaculture activities (Oyarzún et al. 2005; Lara et al. 2009); and erosion control and organic matter accumulation services, which are important for the soil productivity and conservation of the ecosystem integrity (Oyarzún et al. 2005; de Groot et al. 2010). The forest landscape of southern Chile has undergone a progressive anthropization in recent decades due to progressive land-use change and land-use intensification, such as pasture expansion for cattle grazing and commercial plantations (CONAF 2006; Echeverría et al. 2006; Nahuelhual et al. 2014). These land-use change have resulted in major impacts on biodiversity, such as changes in plant species composition, richness of birds and a significant loss of wildlife (Vergara and Simonetti 2004; Echeverría et al. 2007; Jaramillo et al. 2007; Rodríguez-Echeverry et al. 2015), and the provision of the ecosystem services water supply and erosion control in different watersheds (Little et al. 2008; Oyarzún et al. 2011). The foregoing has been the focus of discussion among governmental environmental entities, forestry companies, and the general community; all have highlighted the need to implement effective conservation strategies (Di Marzio and McInnes 2005; CONAF 2006).

We assessed the impact of land-use change and land-use intensification on the spatial patterns of the diversity of native forest habitat, and the influence of these changes on the provision of the following ecosystem services: water supply, control of erosion, and organic matter accumulation in the Río Cruces watershed in southern Chile. We hypothesize that human-induced land-use change has generated diversity loss of native forest habitat, and as a consequence, this loss has caused a decrease in the provision of the ecosystem services.

Methods

Study area

The Río Cruces watershed is located in the Los Ríos region, in southern Chile (Fig. 1). It is located

between the Andes and coast mountain ranges (39°17' S–39°50'S), to the north of the city of Valdivia. The watershed occupies 3640 km² and is characterized by a temperate climate, has a maximum elevation of 826 m, a mean temperature of 12 °C and an annual rainfall of 2293 mm (DGA 2004). Currently, the total area of watershed consist of grassland 31%, native forest habitat 29%, exotic species forest plantations 21%, shrubland 16% and other land-use types (wetlands, bare soil, urban area and bodies of water) 3%. The watershed has a human population of approximately 206,000, which is equivalent to a population density of 46 people km⁻² (DGA 2004). The forestry, agriculture and livestock farming are the main economic activities in the watershed (DGA 2004).

Each of the 262 subwatersheds, which (ranging 500–4000 ha in size), were defined as the spatial unit of analysis. The delimitation of the subwatersheds and all spatial analysis in the study was carried out using the ArcGIS 9.3 (ESRI 2009).

Biodiversity analysis

We analyzed the diversity of native forest habitats as a proxy for biodiversity at the landscape level. This diversity was determined by the variety and abundance of the native forest habitats. The different types of habitat were determined by the presence of native forest in different vegetation types, climatic zones and soil types present in the study area. Vegetation types are areas characterized by plant communities that have a specific physiognomy and species dominance (Luebert and Pliscoff 2004). Climatic zones determine the presence of vegetation types in the landscape. These zones are defined by variables such as precipitation, moisture of air and temperature (Schlatter et al. 1995). Soil types determine the establishment and development of the types of plant species, due to physical–chemical properties and morphological characteristics of the soil (FAO-UNESCO 1971; CIREN 2003). Native forest habitats mapping was carried out using the following maps: (1) Map of native forest, which was extracted from land-use map for the years 1986, 2001 and 2011 (Echeverría et al. unpublished results). In these maps, the following categories of land-use were identified: native forest, exotic species forest plantations, shrubland, grassland, wetland, and other uses (bare soil,

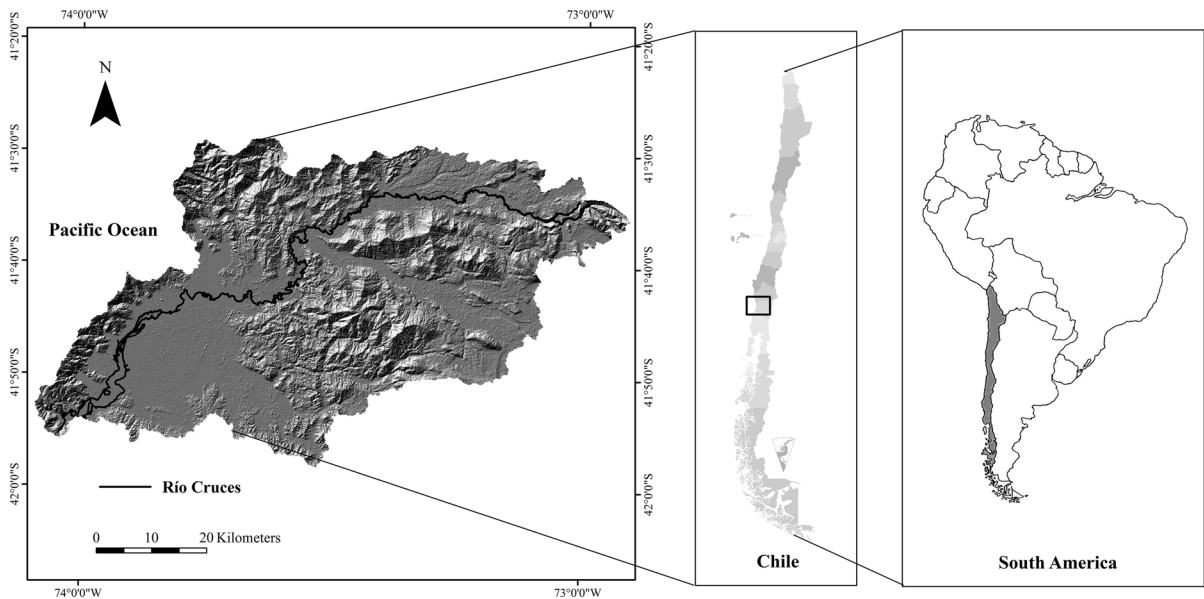


Fig. 1 Location of the Río Cruces watershed, Chile

urban area and water bodies); (2) Map of vegetation types by Estudio de Clasificación de Pisos de Vegetación (Classification Study the Types of Vegetation) (Luebert and Plissock 2004); (3) Map of climatic zones provided by Sistema de Ordenamiento de la Tierra (Land Management System) (Schlatter et al. 1995); (4) Map of soil types provided by Estudio Agrológico de Suelos de Chile (Soil Agrolological Study of Chile) (CIREN 2003). The map of native forest habitats was derived from the calculation from the overlap of these maps. Diversity of the native forest habitats were assessed through Shannon diversity index, which is a landscape metric that relates the variety and abundance of different habitat types. The Shannon diversity index was used as a measure of relative amount of each habitat type in each unit of analysis. The index is expressed by the following formula [1] (McGarigal et al. 2002):

$$H = - \sum_{i=1}^S [(P_i) \times (\ln P_i)] \quad (1)$$

where P_i is the proportion of the landscape occupied by habitat type “i”. The analysis of Shannon diversity index was carried out using FRAGSTATS (McGarigal et al. 2002). Map of native forest habitats were entered into FRAGSTATS and values of the Shannon diversity index were obtained for each subwatershed. All maps used in the study (input and output) had a

spatial resolution of 30×30 m pixels, which is a resolution appropriate for the spatial analysis (Daly 2006).

We assessed the impact of land-use change on the spatial patterns of the diversity of native forest habitat by determining the changes of total area, total number of patches, index richness of habitat patches (the number of different types of habitat patches) and Shannon diversity index. The analysis of these indices was carried out using FRAGSTATS (McGarigal et al. 2002) and the Spatial Analyst extension for ArcGIS 9.3 (ESRI 2009).

N-spect model and ecosystem services

The N-Spect software (No Point Source Pollution and Erosion Comparison Tools) was used to map and analyze the provision of the ecosystem services water supply and erosion control. This software was developed by The National Oceanic and Atmospheric Administration of the United States, and is used as an extension of ArcGIS software. N-Spect was developed to analyze and predict soil erosion through the amount of sediments exported and the potential impacts on water quality from nonpoint sources of pollution (NOAA 2009). It is a software that serves as a tool for management and decision-making concerning water resources, land-use planning,

agricultural policies and practices. N-Spect is a spatially explicit model that uses spatial elevation data to calculate flow direction and flow accumulation throughout a watershed. This software uses data on land cover, types of soils and precipitation to estimate runoff volume obtained after the filtering and retention water. N-Spect also uses the Revised Universal Soil Loss Equation to calculate erosion. The outputs of this software are maps that register the estimates of cumulative runoff and amount of sediment (NOAA 2009).

The parameterization of N-Spect was carried out taking into account the following inputs: (1) Digital elevation model (DEMs) 30×30 m, which was obtained from the Shuttle Radar Topography Mission (SRTM); (2) Map of land-use, in which the following land-use categories were identified: native forest, exotic species forest plantations, shrubland, grassland, wetland, and other uses (bare soil, urban area and water bodies); (3) Maps of annual precipitation and rainfall erosivity (R factor), which combine the effect of the duration, magnitude and intensity of each rainfall event. The average annual rainfall erosivity were calculated from daily rainfall data registered in 18 meteorological stations located in the study area, using the method and formula [2] proposed by Angulo-Martínez and Beguería (2009):

$$EI = EI_{30} = \left(\sum_{r=1}^0 e_r v_r \right) I_{30} \quad (2)$$

where e_r and v_r are, respectively, the unit rainfall energy ($\text{MJ ha}^{-1} \text{mm}^{-1}$) and the rainfall volume (mm) during a time period r , and I_{30} is the maximum rainfall intensity in a 30 min period during the event (mm h^{-1}). The unit rainfall energy (e_r) is calculated for each time interval by the following formula [3] Angulo-Martínez and Beguería (2009):

$$e_r = 0.29[1 - 0.72 \exp(-0.05i_r)] \quad (3)$$

where i_r is the rainfall intensity during the time interval (mm h^{-1}). These annual averages were spatialized using geostatistical method of topoclimatological interpolation (Díaz et al. 2010); (4) Values of vegetation coverage (C factor), which reflects the effect of different crops and management practices on erosion rates. These values were estimated on the basis of map of land-use and values proposed by Wischmeier and Smith (1978); (5) Coefficient of soil

erodibility (K factor), which depend on the texture, structure and content of organic matter that has the soil types. These coefficients were calculated using information from the Estudio Agrológico de Suelos de Chile (CIREN 2003), using the equation of the nomogram proposed by Wischmeier and Smith (1978); (6) Values for hydrological groups for each soil types; these values are related to the number curve method that is based in the permeability of different soil types (NOAA 2009). The values from the hydrological groups were estimated from soil texture data (CIREN 2003). Information on the different soil types that form part of the study area was entered into a map, which was provided in the Estudio Agrológico de Suelos de Chile (CIREN 2003).

Model validation was carried out with the data registered in four precipitation and sediment stations of the General Directorate of Water present in the study area. The goodness of model fit was assessed by the method of quantitative assessment of “relative efficiency” proposed by Thanapakpawin et al. (2007). The relative efficiency is expressed by the following formula [4]:

$$Erel = 1 - \frac{\sum_{i=1}^N \left(\frac{C_i - C'_i}{C_i} \right)^2}{\sum_{i=1}^N \left(\frac{C_i - \hat{C}}{\hat{C}} \right)^2} \quad (4)$$

where C_i is observed discharge, C'_i is the simulated discharge, \hat{C} is the mean observed discharge, and N is the total number of samples. *Erel* measures the goodness of model fit by comparing the volume of the discharge. The difference between simulated and observed values was quantified using relative deviations instead of absolute values (Thanapakpawin et al. 2007).

Water Supply This ecosystem services is the volume of water produced (m^3/ha) per subwatershed (de Groot et al. 2010). This service is the water potentially available to humans (Chan et al. 2006). Water supply refers to the runoff volume obtained after filtering and retention water (de Groot et al. 2010). The filtering is mainly performed by the vegetation coverage and soil. The retention capacity depends on topography and subsurface characteristics of the landscape. This services was modeled by N-Spect using precipitation, land cover, soil and digital elevation model as parameters.

Erosion Control This ecosystem services is the ability of natural vegetation to curb erosion by holding onto soil (Egoh et al. 2009), which is measured as the amount (tons/ha) of sediment exported from each subwatershed (de Groot et al. 2010). Soil erosion removes nutrients and reduces fertility (de Groot et al. 2010), and may generate sedimentation and eutrophication of nearby rivers (Egoh et al. 2009). Therefore, areas where vegetation holds the soil need to be managed to allow continuous delivery of multiple services (de Groot et al. 2010). In this study the erosion control services were modeled using the amount of sediment exported. This services was modeled by N-Spect using the Revised Universal Soil Loss Equation.

Organic Matter Accumulation This ecosystem service is directly linked to the accumulation of

organic matter in the soil (Yuan et al. 2006; Egoh et al. 2009). Experts in this area have registered a positive correlation between soil depth and vegetation coverage with the organic matter present in the soil (Yuan et al. 2006). Accordingly, these two variables have been used for modeling soil formation (Yuan et al. 2006). In this study, the soil depth and coverage of different native forest habitats were used as proxies for organic matter accumulation. Values of depth of the soil types were obtained from Estudio Agrológico de Suelos de Chile (CIREN 2003). The thresholds of depth ≤ 1 m (slightly depth) and > 1 m (depth) were used in the modeling; these thresholds were based on the literature (CIREN 2003). Values of coverage of different native forest habitats were obtained from the map of native forest habitats. This service was modeled using the index of organic

Table 1 Habitat types of the Valdivian Temperate Forest in the Río Cruces watershed

Habitat type	Variables	Climatic zone	Soil types
	Vegetation types		
I	Andean temperate deciduous forest of <i>Nothofagus alpina</i> and <i>Dasyphyllum diacanthoides</i>	Zone 2, district 0	Andisol
II	Andean temperate deciduous forest of <i>Nothofagus alpina</i> and <i>Nothofagus dombeyi</i>	Zone 2, district 0	Andisol
III	Temperate deciduous forest of <i>Nothofagus obliqua</i> and <i>Laurelia sempervirens</i>	Zone 1, district 0	Andisol
IV	Temperate deciduous forest of <i>Nothofagus obliqua</i> and <i>Laurelia sempervirens</i>	Zone 1, district 0	Ultisol
V	Temperate deciduous forest of <i>Nothofagus obliqua</i> and <i>Laurelia sempervirens</i>	Zone 2, district 0	Andisol
VI	Temperate deciduous forest of <i>Nothofagus obliqua</i> and <i>Laurelia sempervirens</i>	Zone 2, district 0	Ultisol
VII	Temperate laurifolio forest of <i>Nothofagus dombeyi</i> and <i>Eucryphia cordifolia</i>	Zone 1, district 0	Andisol
VIII	Temperate laurifolio forest of <i>Nothofagus dombeyi</i> and <i>Eucryphia cordifolia</i>	Zone 1, district 0	Ultisol
IX	Temperate laurifolio forest of <i>Nothofagus dombeyi</i> and <i>Eucryphia cordifolia</i>	Zone 2, district 0	Andisol
X	Temperate laurifolio forest of <i>Nothofagus dombeyi</i> and <i>Eucryphia cordifolia</i>	Zone 2, district 0	Ultisol

Characteristics of climatic zones and soil types

Climatic zone 1, district 0: Total annual precipitation (mm): 1900 min–2000 max. Annual moisture index 2.0 min–2.5 max. Dry period 1–2 months/year. Frost-free period 200–250 days/year. Total number of frost 10–20 days/year. Estival moisture index 0.5 min–0.6 max. Average relative humidity in January 70–80%. Annual absolute temperature min – 6 °C, frequency of occurrence 1 month/year

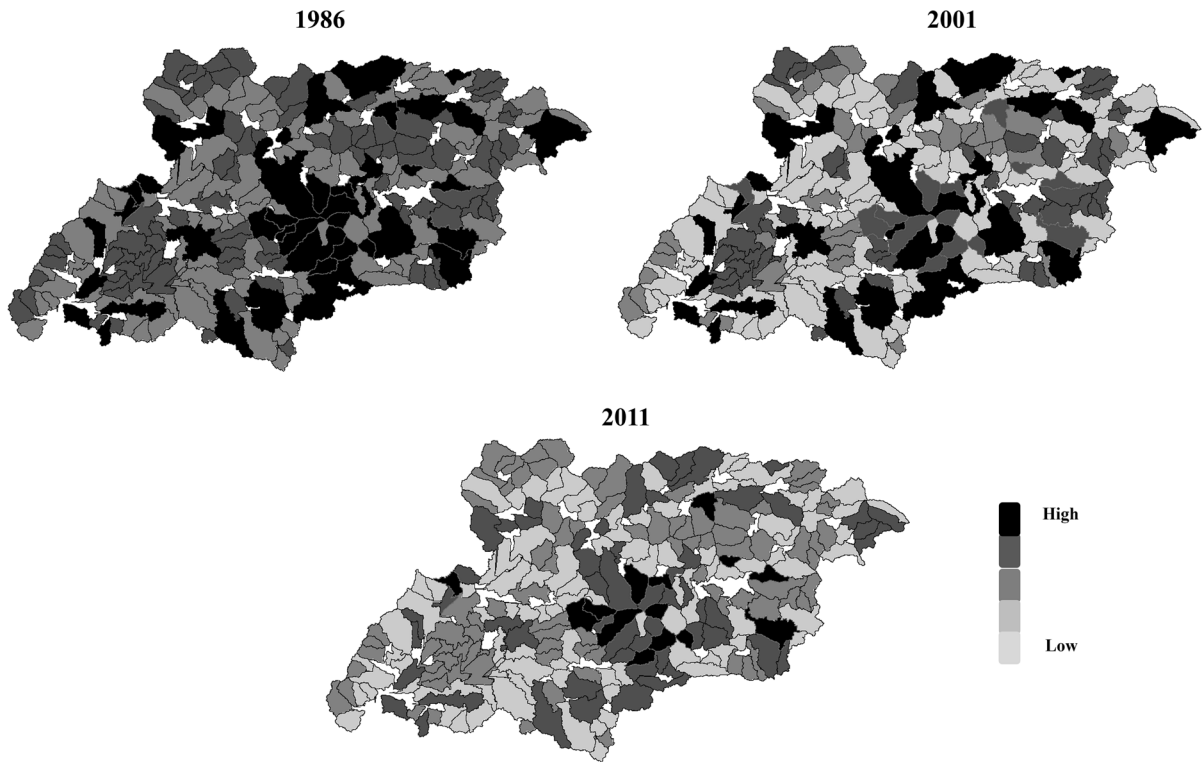
Climatic zone 2, district 0: Total annual precipitation (mm): 1900 min–3000 max. Annual moisture index 2.0 min–2.5 max. Dry period 1–2 months/year. Frost-free period 120–200 days/year. Total number of frost 20–30 days/year. Estival moisture index 0.5 min–0.6 max. Average relative humidity in January 65–70%. Annual absolute temperature – 6 °C, frequency of occurrence 2 month/year

Andisol type: Soil derived from volcanic ash. These soils in Chile correspond to Trumaos and Ñadis soils. Andisol soils have excellent physical and morphological conditions, whereby can be grown easily. These soils have large amounts of phosphorus but it is retained in the soil in a form that is not available to plants. Therefore, these soils require large amounts of phosphatic fertilizations to obtain high yields

Ultisol type: Soils with B Horizon well expressed due to an increase of clay in the A horizon. These soils are highly leached. Consequently, it has low levels of nutrients. These soils require large amounts of fertilization to obtain reasonable yields

Table 2 Area of the habitat types of native forest and index richness of habitat patches in the Río Cruces watershed

Year	Area (ha) of the different habitat types										Index richness of habitat patches
	I	II	III	IV	V	VI	VII	VIII	IX	X	
1986	2816	860	794	5093	8391	10,453	122	1783	2034	5143	2.55
2001	2098	842	791	4844	8058	9722	88	1278	1384	4517	2.53
2011	2013	836	771	4714	7990	9624	83	1227	1319	4507	2.53

**Fig. 2** Biodiversity in the Río Cruces watershed, Chile

matter accumulation (Egoh et al. 2009). This index was calculated for each subwatershed by weighted the areas of vegetation coverage and the areas of the different types of soils, and then dividing this value by the total area of the corresponding subwatershed. This index was spatialized for each subwatershed through the ArcGIS.

Generalized linear models (GLMs)

The relationship between changes in the spatial patterns of the diversity of native forest habitat and provision of the ecosystem services was analyzed

using GLMs. These models are extensions of linear models that allow for greater flexibility than Gaussian linear models (Zuur et al. 2009; Hossain et al. 2015). The provisions of the ecosystem services were used as dependent variables, while the changes of the following spatial patterns: total area, total number of patches, index richness of habitat patches and Shannon diversity index were used as independent variables. GLMs were carried out using R statistical software (version 3.0.1) (Venables et al. 2013). In these models the test distribution χ^2 and the function “drop1” were used to assess statistical significance of each of the variables. In order to find the most

effective model, the variables of less significance were removed (Hossain et al. 2015).

Results

In the study landscape, ten types of native forest habitats were recorded (Table 1). These were determined by the presence of the Valdivian Temperate Forest in ten vegetation levels, two climatic zones and two soil types (Table 1). The ten types of native forest habitats had an area that ranged from 122 to 10,453 ha in 1986 and 83 to 9624 ha 2011 (Table 2). The Shannon diversity index ranged from 0.71 to 0.51 over the entire study period (Fig. 2).

There was an area loss of 12% of native forest habitat from 1986 to 2011 (i.e. a mean annual loss of 0.5%) (Figs. 3, 4). The largest area loss (10.3%) occurred from 1986 to 2001 (i.e. a mean annual loss of 0.73%) (Figs. 3, 4). In 1986 there were 17,031 patches of native forest habitat, with a total area of 37,490 ha (Figs. 3, 4). In 2011 the number of patches

Fig. 4 Relationship among area loss of native forest habitat, increasing number of habitat patches and diversity loss with provision of the ecosystem services: **a** erosion control and water supply; **b** organic matter accumulation in the Río Cruces watershed

increased to 26,352, with a reduction to a total area of 33,084 ha. The largest increase in the number of patches (13,269) occurred from 1986 to 2001. In 2011 the diversity of native forest habitat was restricted to small patches sparsely distributed across of the landscape (Fig. 3).

There was an important decrease of 0.20 in the Shannon diversity index over the entire study period (Fig. 4a). The largest decrease in the Shannon diversity index (0.17) occurred from 1986 to 2001 (Fig. 4a). There was a slight decrease of 0.02 in the index richness of habitat patches from 1986 to 2001 (Table 2). There was no change in the index richness over the second study period.

The modeling of the ecosystem services carried out in the N-Spect software registered a high

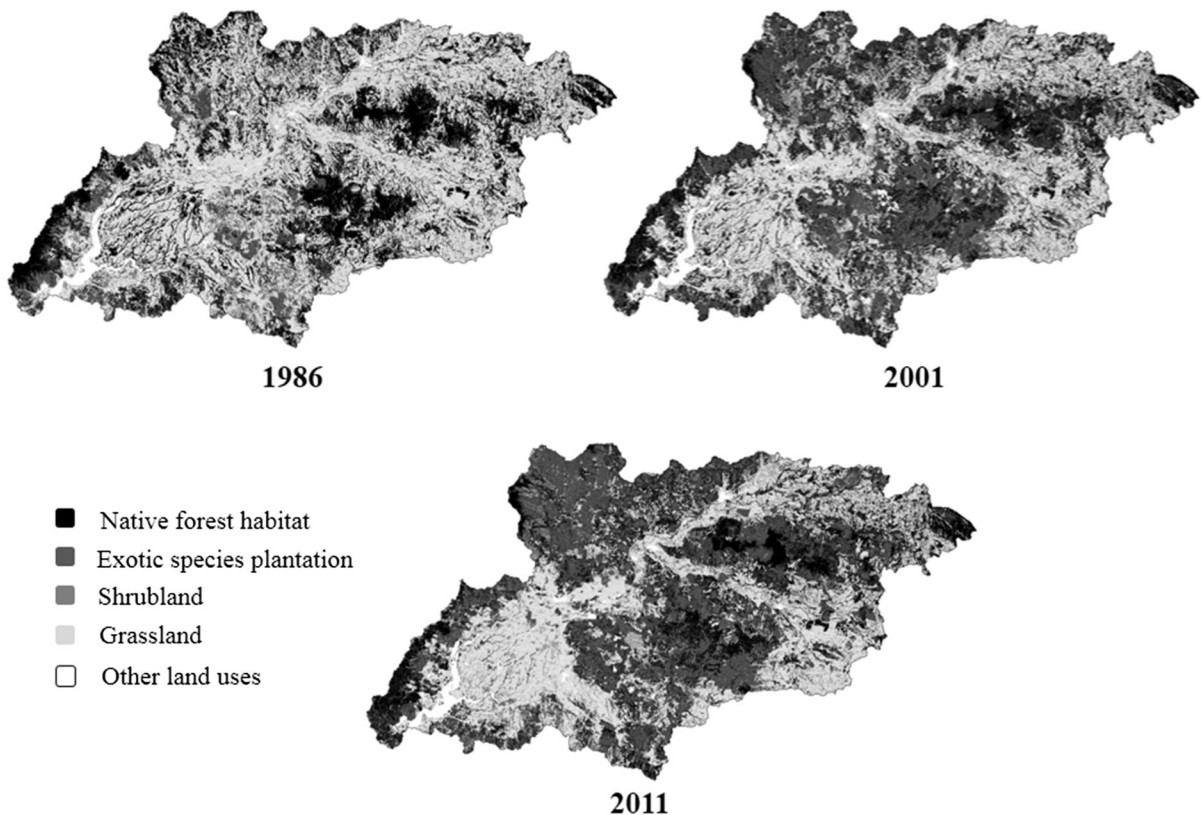
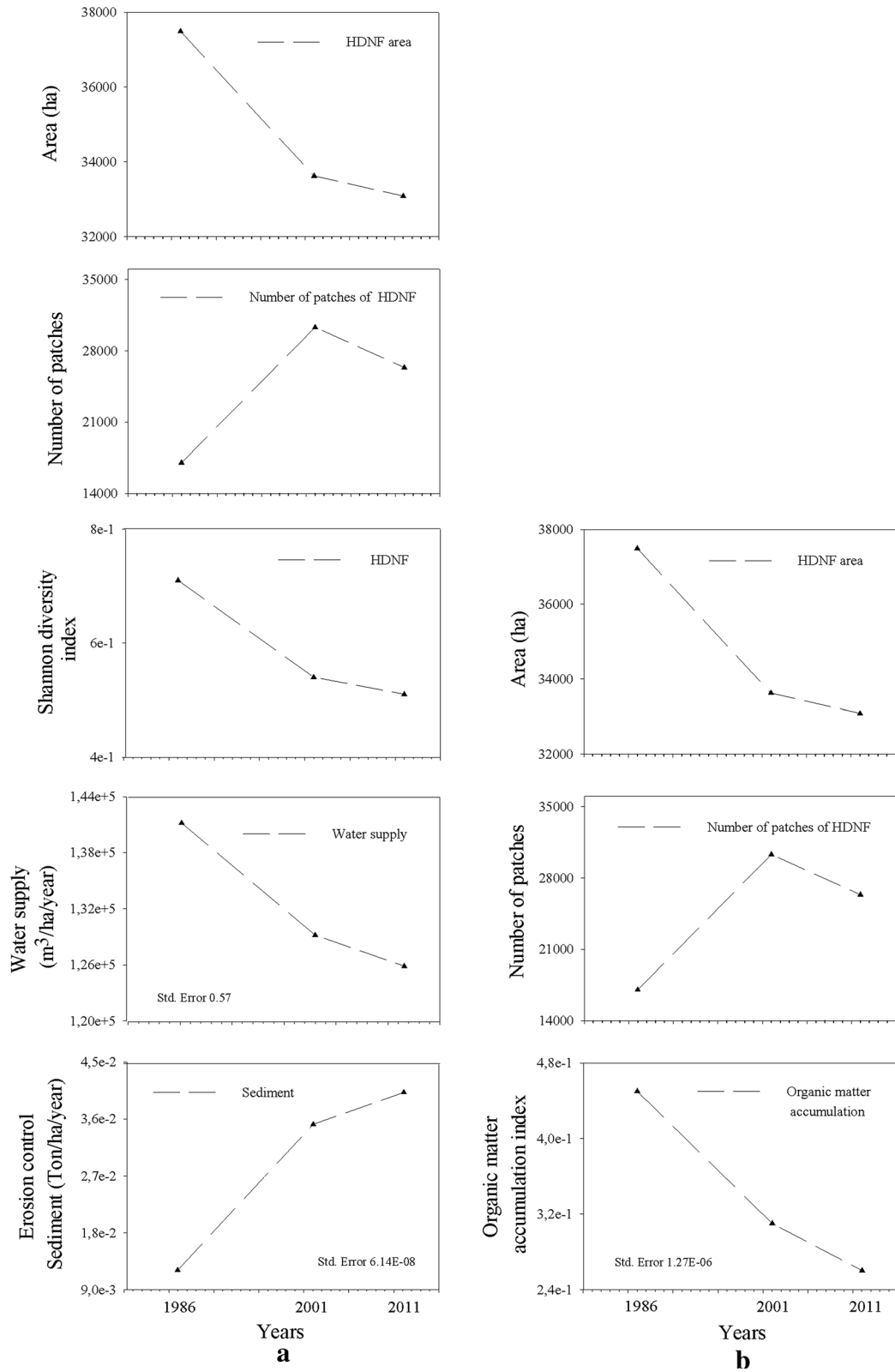


Fig. 3 Patterns of land-use change in the Río Cruces watershed from 1986 to 2011



accuracy. The modeling of the ecosystem service water supply registered an efficiency of 0.93, and that of the ecosystem service erosion control registered an efficiency of 0.95. These results are a reliable representation of hydrologic processes and sediment loads. The models accuracy reveals that N-Spect is a versatile software that integrates multiple environmental variables that can support the effective watershed management.

There were decreases of 346, 41 and 11% in the provision of the ecosystem services erosion control, organic matter accumulation and water supply, respectively from 1986 to 2011 (Figs. 4, 5). The largest decreases in the provision of the services (312, 29 and 9% for erosion control, organic matter accumulation and water supply services, respectively) were registered from 1986 to 2001.

The decrease in the provision of the services water supply and erosion control can be mainly explained by the interaction between area loss of native forest habitat, increase in number of patches of native forest habitat and decrease in the Shannon diversity index ($p < 0.001$) (Table 3 and Fig. 4a). The decrease in the provision of the organic matter accumulation service can be explained by the interaction between area loss of native forest habitat and the increasing number of patches ($p < 0.01$) (Table 3 and Fig. 4b). The moderate change registered in the index richness of habitat patches did not explain the decrease in the provision of the ecosystem services.

Discussion

Changes in the diversity of native forest habitat

Our results indicate that there was a substantial loss of native forest habitat in the study landscape from 1986 to 2011, although the annual rate of loss (0.5%) was lower than that recorded for other hotspot of habitat diversity that has been significantly transformed, such as the Lancang River Valley in the southern China (5.5% annual loss; Liu et al. 2014) and the Dorset County on the south coast of England (2% annual loss; Hoofman and Bullock 2012). The loss of native forest habitat was associated with an important increase in the number of patches, from 17,031 in 1986 to 26,352 in 2011, which is one of the main symptoms of fragmentation (Lindenmayer

2016). If the current trajectory of loss and fragmentation of native forest habitat continues, further losses of diversity and richness of habitat patches can be expected.

Our results show that there was a substantial and important diversity loss (0.20) from 1986 to 2011. This trend in diversity loss is similar to that reported for other forest landscapes where the habitat diversity has undergone a significant transformation (Benini et al. 2010; YuhaiBao et al. 2011; Wan et al. 2015). In the study landscape, the diversity loss was associated with a slight loss of richness of habitat patches, both losses were due to the loss and fragmentation of native forest habitat over the entire study period. The foregoing demonstrates that main impact of land-use change and land-use intensification was on the abundance of each habitat type, but not the richness of these. In other words, it is likely that different types of native habitats have remained, but the transformed areas were used intensively and that is what has led to loss of diversity of native forest habitats. Our study indicates that if current trajectory of diversity loss continues, this could generate alterations in the provision of the ecosystem services.

N-Spect model accuracy

The high accuracy of the model (> 0.93) shows that N-Spect is a powerful tool for the spatial evaluation of the provision of the ecosystem services. The N-Spect software model integrated a variety of information from the physical environment, and based on that data an innovative analysis on ecosystem services at landscape level was performed. This study demonstrates that N-Spect has the potential to provide references to ecology planners and local stakeholders to address landscapes issues at different spatial scales. Since the validation of N-Spect models was carried out with data recorded in the four precipitation and sediment stations present in the study area, the validation may have a small inaccuracy. Therefore, it is necessary that future studies use a larger number of stations to obtain greater accuracy in the modeling.

Fig. 5 Mapping of the ecosystem services **a** water supply, **b** erosion control and **c** organic matter accumulation in the Río Cruces watershed between 1986 and 2011

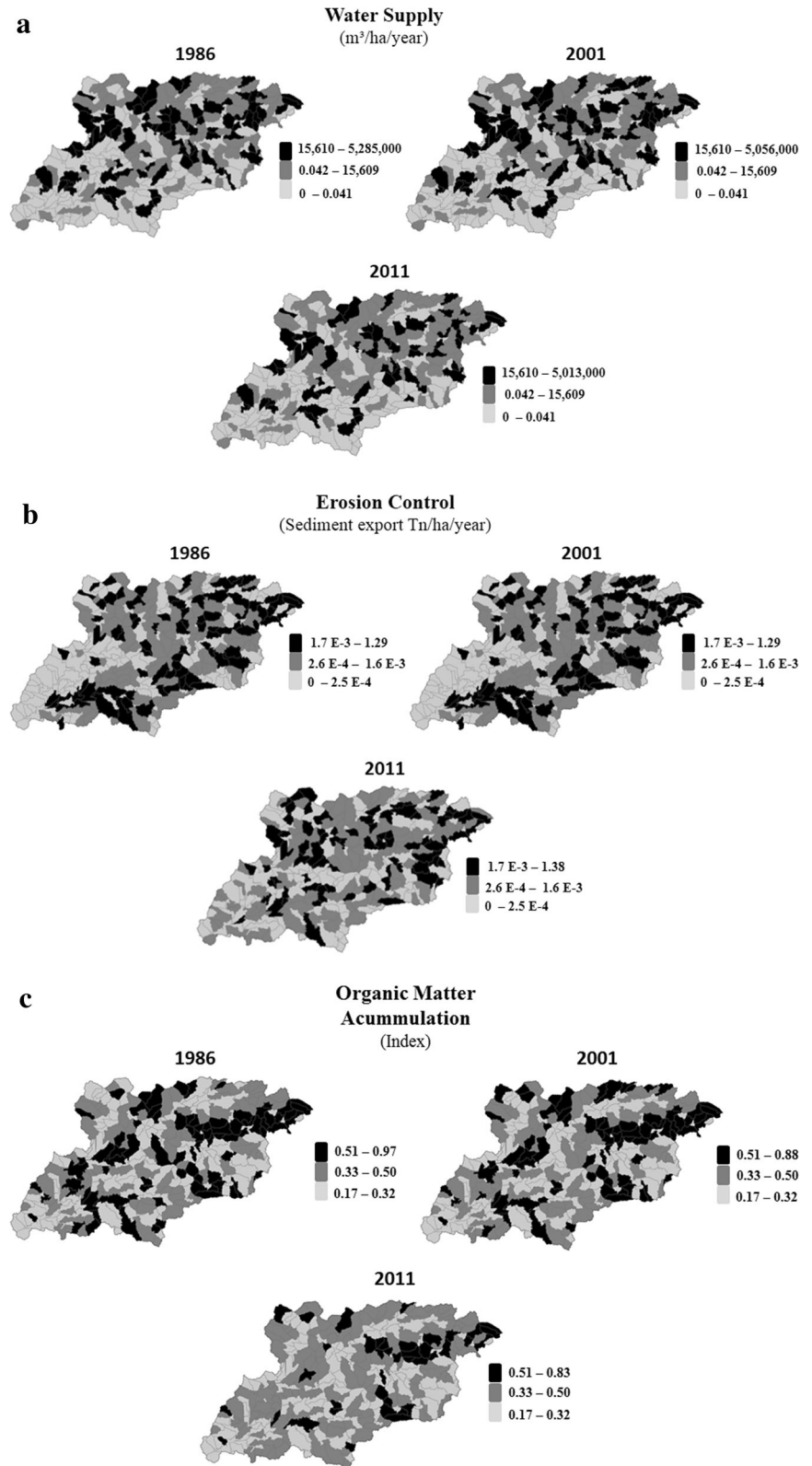


Table 3 Generalized linear models (GLMs) built based on the interaction of the changes in the spatial patterns of the diversity of native forest habitat, which explain the loss of provision of the ecosystem services in the Río Cruces watershed from 1986 to 2011

Variables	Estimate	Std. Error	T value	<i>p</i> value
Water supply (intercept)	8235.102	7335.192	0.874	0.345
Area: diversity loss	− 531.545	104.578	− 4.731	< 0.001
Increase in number of patches: diversity loss	271.352	117.278	2.350	0.010
Area loss: increase in number of patches: diversity loss	− 2.127	0.498	− 3.508	< 0.001
Erosion control (intercept)	− 0.003	0.001	− 2.890	0.001
Area loss: diversity loss	< 0.001	< 0.001	1.282	0.025
Area loss: increase in number of patches: diversity loss	< 0.001	< 0.001	1.870	0.026
Organic matter accumulation (intercept)	0.227	0.040	4.745	< 0.001
Area loss: increase patches number	< 0.001	< 0.001	1.989	0.004

$N = 262$; $p < 0.05$; $p < 0.01$; $p < 0.001$

Changes in the provision of the ecosystem services

Our study shows that the changes in the spatial patterns of the diversity of native forest habitat were strongly linked to a decrease in the provision of the ecosystem services from 1986 to 2011. As loss and fragmentation of native forest habitat and the diversity loss increases, the provision of ecosystem services decreases. A similar relationship has been reported in other hotspot landscapes that have undergone different anthropogenic interventions (Zhao et al. 2006; Qi et al. 2014). In the study landscape, the loss of provision of the organic matter accumulation service was significantly related with the loss and fragmentation of native forest habitat. The foregoing probably due to the importance of the abundance of the native forest habitats in the accumulation of organic matter (de Groot et al. 2002; Egoh et al. 2009). In contrast, the losses of the erosion control and water supply services registered a significant relationship with the loss and fragmentation of native forest habitat and diversity. This loss and fragmentation is possibly due to the importance of the abundance and richness of type of native forest habitats in: (a) the retention of the precipitation and the gradual delivery of it in the water network (Lara et al. 2009) and (b) the retention of sediments carried out by the roots of the variety of plants (van Jaarsveld et al. 2005; Egoh et al. 2009) The richness of type of native forest habitat has a key role in the provision of erosion control and water supply services, probably

due to the ecological function that plant communities perform in each habitat. It is important to consider that a decrease of 8.5% in precipitation was recorded by the Dirección General de Agua over the entire study period (DGA 2014). Therefore, the decrease of water supply services is not only influenced by different impacts of anthropogenic land-use change, but also by the variability of precipitation. Our results show that the provision of the ecosystem services depends on the abundance and/or richness of type of native forest habitats. Our study also indicates that the loss in provision of each service occurs differently, depending on the interaction of the different impacts of the land-use change on the diversity of native forest habitat.

Conclusions

Our study shows that the loss of provision of the ecosystem services is strongly linked to the loss of diversity of native forest habitat. These losses highlight the need to establish conservation priority of the landscape studied. We recommend a landscape approach, using the corridor–patch–matrix model (Lindenmayer and Franklin 2002) as a conservation planning strategy. This model is appropriate given the current configuration of the landscape, with native forest patches sparsely distributed across the landscape. The main objective of the corridor–patch–matrix model is to maintain the quality and quantity of patches of native forest habitat through

management of the matrix (Lindenmayer and Franklin 2002). In the study landscape, the management of the matrix should focus on sensitive buffer areas that improve the connectivity between patches of native forest habitat, which can maintain and increase the abundance and richness of these, and the provision of multiple ecosystem services. We recommend that this conservation strategy be complemented with land-use planning focused on sustainable production practices, such as agroforestry. The design and implementation of such a strategy requires identification and evaluation of sensitive buffer areas and their connectivity. Moreover, this strategy must be supported by a framework of environmental policies, which must be based on the 17 UN sustainability criteria and generated by the Chilean government.

Finally, studies that assess the spatial congruence between the diversity of native forest habitat and ecosystem services are needed. These studies should provide a better understanding to what extent biodiversity conservation ensures the provision of multiple services, which will contribute in the implement simultaneous conservation of both resources and will assist in practical decision making.

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