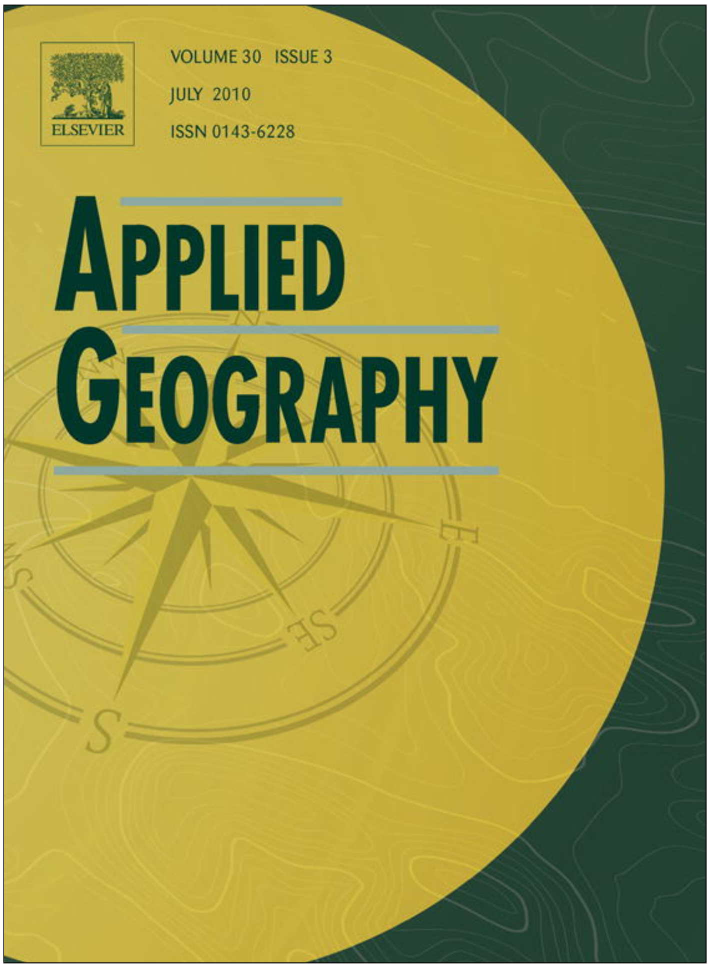
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Monitoring land cover change of the dryland forest landscape of Central Chile (1975–2008)

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A B S T R A C T

Land cover and its configuration in the landscape are crucial components in the provision of biodiversity and ecosystem services. In Mediterranean regions, natural landscapes mostly covered by evergreen vegetation have been to a large extent transformed into cultural landscapes since long time ago. We investigated land cover changes in Central Chile using multi-temporal satellite imagery taken in 1975, 1985, 1999 and 2008. The major trends in this highly dynamic landscape were reduction of dryland forest and conversion of shrubland to intensive land uses such as farmland. The average net annual deforestation rate was −1.7%, and shrubland reduction occurred at an annual rate of −0.7%; agriculture, urban areas and timber plantations increased at annual rates of 1.1%, 2.7% and 3.2%, respectively, during the 1975–2008 period. Total forest and shrubland loss rates were partly offset by passive revegetation. However, most of the areas that were passively revegetated remained as shrubland and did not turn into forests due to a low capacity of forest recovery. This resulted in a progressive loss and degradation of dryland forest over the entire region. Overall, the documented land cover changes increase provisioning services such as crops, cattle, and timber that are characteristic of cultural landscapes in the area but may cause an irreversible loss of biodiversity and a depletion of other ecological services provided by forests and shrubland. The implications for conservation of this area and the need for territorial planning and adapted land-use strategies are discussed.

Introduction

Natural landscapes, i.e. those unaffected or hardly affected by human activities, are being transformed into cultural landscapes throughout the world (Feranec, Jaffrain, Soukup, & Hazeu, 2010; Foley et al., 2005; López & Sierra, 2010). This transformation trades off the biodiversity and ecosystem services which are characteristic of both types of landscapes, e.g. higher levels of biodiversity and supporting and regulating services in natural landscapes vs. higher levels of provisioning services such as crop and timber production in cultural landscapes (Millennium Ecosystem Assessment, 2005; Rey Benayas,

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Newton, Díaz, & Bullock, 2009). As the characteristics of land cover have important impacts on climate, biogeochemistry, hydrology and species diversity, land cover change has been indicated as one of the high priority concerns for research and for the development of strategies for sustainable management (Turner, Moss, & Skole, 1993; Vitousek, 1994). In recent years, special attention has been given to land-use changes and dryland degradation (Reynolds, Maestre, Kemp, Stafford-Smith, & Lambin, 2007). Vegetation cover in these ecosystems with limited primary productivity plays a crucial role in providing services such as climate and water regulation (Millennium Ecosystem Assessment, 2005).

Mediterranean ecosystems are a particular type of dryland that account for less than 5% of the Earth's surface but host about 20% of the world's plant species, many of which are endemic (Cowling, Rundel, Lamont, Kalin Arroyo, & Arianoutsou, 1996). Land degradation, i.e. the substantial decrease in the biological productivity of the land system, resulting from human activities rather than natural events (Johnson & Lewis, 2006) is an important issue in many Mediterranean regions (Conacher & Sala, 1998; Geri, Amici, & Rocchini, 2010). Loss of natural vegetation cover is often a precedent to soil erosion and deterioration of the water storage capacity; these modifications of the land system may lead to desertification due to longer term factors such as climate change, triggering short term degradation of ecosystems by humans (Reynolds & Stafford Smith, 2002). Nevertheless, in the Mediterranean basin and California, the loss of vegetation cover has been partly counterbalanced by vegetation recovery over the last decade (Carmel & Flather, 2004; Lasanta, González-Hidalgo, Vicente-Serrano, & Sferi, 2006; Mouillot, Ratte, Joffre, Mouillot, & Rambal, 2005; Pueyo & Beguería, 2007; Romero-Calcerrada & Perry, 2004), which occurred mostly due to concentration of crop production and abandonment of less productive farmland. The polarisation between more intensive and more extensive use of land has been described as the main trend of current landscape changes (Antrop, 2005).

In Mediterranean Central Chile, however, this trend is less clear. Whereas land abandonment is occurring in some areas as a result of soil degradation, threats to sclerophyllous forests and shrublands, such as urban and agricultural expansion, cattle grazing, logging for firewood, and introduction of alien species, still persist throughout the region. Some studies have described vegetation degradation in Central Chile concerning disturbances of successional trajectories at a rather local scale (e.g. Balduzzi, Tomaselli, Serey, & Villaseñor, 1982; Fuentes, Avilés, & Segura, 1989; Holmgren, 2002) and pressures on vegetation due to land occupation patterns (Ovalle, Avendaño, Aronson, & Del Pozo, 1996). Common patterns of landscape change throughout Central Chile, including the description of severe reduction of natural vegetation, have been described rather qualitatively (e.g. Armesto, Arroyo, Mary, & Hinojosa, 2007; Aronson et al., 1998). So far, these processes have not been mapped and quantified at a regional scale, and change trajectories among land cover types have not been systematically evaluated and explained.

To address the issue of gaining a systematic understanding of the magnitude of land cover changes at the regional scale we considered the advantages of remote sensing data to detect, measure and monitor land cover change due to this system's ability to capture an instantaneous synoptic view of a large part of the Earth's surface and acquire repeated measurements of the same area on a regular basis (Donoghue, 2002). Land cover detection and monitoring are especially useful in those regions where there is a lack of available cartographic information with sufficient spatial resolution to examine how humans change land cover and to provide a basis for conservation and restoration planning. To our knowledge, this is the first study that has explored the recent historical and current extent of land cover types, as well as the changes that have occurred in Central Chile over a 33-year period (1975–2008). The main goal of this study is to investigate the dynamics of land cover change, focussing on the dynamics of natural vegetation cover as a result of land-use pressure, particularly a hypothesised expansion of cropland, pastures and timber plantations. The specific scope of this paper is embedded in the dimension of land change science that focuses on observation, monitoring, and land characterization (Turner, Lambin, & Reenberg, 2007). We specifically address: (1) area change and change rate; (2) spatial distribution of changes; (3) change trajectories of land cover types; and (4) accuracy assessment of change detection. The information generated in this study will be a useful basis for analyzing underlying causes of change and designing management strategies, as it identifies the spatio-temporal patterns associated with landscape processes that might affect policy making, conservation and restoration programs.

Material and methods

Study area

The study area is located in the Mediterranean bioclimatic zone of Central Chile (Amigo & Ramírez, 1998) and extends over 13,175 km^2, covering parts of the Valparaíso, Libertador Bernando O'Higgins and Metropolitan administrative regions (Fig. 1). The area includes characteristic landscapes of the Mediterranean zone, like parts of the coastal plains, the coastal mountain range and the Central Valley. The rationale for defining the boundaries of the study area based on the bioclimatic zone was that vegetation has a similar response to biotic and abiotic factors. For example, vegetation recovery is constrained within this bioclimatic area by water availability. Additional criteria used for defining the study area were that it shares a relatively common pattern of land use and that it concentrates a large population that may put high pressure on natural resources.

Altogether the area is characterised by dry summers and wet winters with strong inter-annual variability due to the El Niño–Southern Oscillation (ENSO) phenomenon. The mean annual temperature is 13.2 °C, and the mean annual precipitation is 531 mm. Temperature and moisture patterns are primarily a function of topography (Badano, Cavieres, Molina-Montenegro, & Quiroz, 2005), with elevations ranging from sea level to 2260 m in the coastal mountain range. The climatic variability and varied topography result in a spatially heterogeneous mosaic of vegetation. At present, *Acacia caven* shrubland

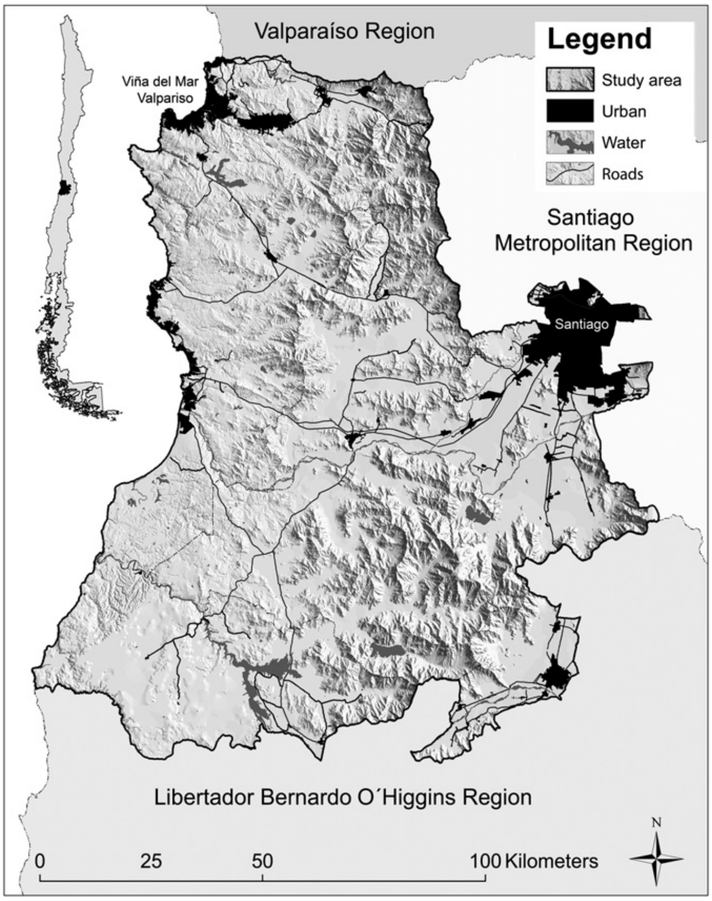


Fig. 1. Location of the study area in Central Chile, between 33°51′00″–34°07′55″ S and 71°22′00″–71°00′48″ W.

is predominant and covers most of the lower hillslopes, whereas evergreen sclerophyllous forest remains in drainage corridors and on steeper slopes with southern aspects.

The pre-Columbian vegetation of Central Chile is described to have been dense and diverse, with a predominance of evergreen sclerophyllous trees and shrubs (Balduzzi et al., 1982). Historical records indicate that Central Chile has experienced profound landscape transformations since the mid-sixteenth century resulting from logging, agriculture expansion and livestock overgrazing (Vogiatzakis, Mannion, & Griffiths, 2006). Land use is mostly concentrated in flat valleys, where the major agricultural activities are vineyard and fruit cultivation as well as corn and wheat cropping. Two of the most important uses of native forest resources by local communities are extraction of fuelwood and extensive livestock husbandry. Conversions to commercial timber plantations with exotic species like *Pinus radiata* and *Eucalyptus globulus* have also occurred in the flat coastal zone since the 1970s (Aronson et al., 1998).

The study area is home to circa 5.2 million inhabitants, which represents around 34% of the Chilean population. The population increased by 53% between 1970 and 2002, and the percentage of urban population as compared to rural population remained high (93% urban population in 1970 and 96% in 2002). The region is acknowledged as one of the world's 25 biodiversity hotspots (Myers, Mittermeier, Mittermeier, da Fonseca, & Kent, 2000) and is home to approximately 2400 plant species, 23% of which are endemic (Cowling et al., 1996).

Multi-temporal land cover classification

Land cover change was evaluated with a post-classification procedure in order to obtain a matrix of change directions among land cover classes (Lu, Mausel, Brondizio, & Moran, 2004). A time series of four pairs of unprocessed Landsat images (path 233, row 83, and path 233, row 84) for the years 1975 (MSS), 1985 (TM), 1999 (ETM+), and 2008 (TM) was used. Each pair comprised two neighbouring scenes from the same date taken under relatively clear sky conditions (<10% cloud cover). Due to a prevailing cloud cover in the southern coastal range of the study area, it was impossible to obtain pairs of images from the same month for the whole 33-year period. To avoid major differences in phenology, all images obtained were taken during the dry season (image dates from November to March) and in La Niña years. As fluctuations in precipitation are relevant for the spectral response of biomass, all selected scenes represent similar drought conditions and correspond closely to mean monthly values throughout the study period. All images were pre-processed, including geometric, atmospheric and topographic corrections (Appendix A).

Table 1
Description of major land cover types defined in this study.

Class	Description
Forest	75–100% canopy cover, advanced stage of succession of sclerophyllous forest with species like *Cryptocarya alba*, *Peumus boldus*, *Quillaja saponaria*, *Lithrea caustica* and deciduous forest, mainly *Nothofagus macrocarpa* and *Ribes punctatum*
Shrubland	25–75% cover of shrub species, such as *Acacia caven*, *Maytenus boaria*, *Prosopis chilensis*, *Trevoa trinervis*, *Colliguaja odorifera*
Agriculture	Rainfed and irrigated agriculture, wine yards, fruit orchards
Urban	Urban and industrial areas
Bareland	Rocks, beach and dunes, bare river beds, permanently degraded land, newly cleared land
Water	Rivers, lakes, water reservoirs
Pasture	Grassland with less than 25% shrub cover
Plantation	Timber plantations with *Pinus* and *Eucalyptus* in advanced growth stage

Nineteen land cover classes were initially defined. Four hundred and ninety-eight field points were taken with a GPS in order to train the spectral signature of the selected land cover classes in a supervised classification scheme. Informal interviews of land owners and land managers were conducted during field survey to obtain information on previous and current land cover and land use. This information was complemented with high-resolution imagery obtained from Google Earth to account for areas with restricted accessibility. To train the classification, a region growing approach with the "seed" function (PCI, 2000) was used. This approach starts with a set of "seed" points and appends neighbouring pixels to the seeds that have same spectral properties. Training areas for the spectral signatures of older images were selected in those sites where land cover remained unchanged or by using areas with similar spectral characteristics.

Signature separability of the initial classes for all images was evaluated using the Bhattacharyya distance. Based on this distance, classes were iteratively merged until reasonably high signature separability was achieved. Specific separability values for forest vs. timber plantations and for urban vs. bareland remained low (values of Bhattacharyya distance >1.3), but these classes were not merged due to their importance for land planning. The iterative process of selecting consistent land cover classes throughout the time series resulted in eight final land cover classes (Table 1). The average separability for all signatures reached a value of Bhattacharyya distance >1.9, indicating good overall separability (PCI, 2000).

Classification of the resulting eight land cover classes was performed using the maximum likelihood algorithm. This procedure has proven to be a robust and consistent classifier for multi-date classifications (e.g. Shalaby & Tateishi, 2007; Wu et al., 2006; Yuan, Sawaya, Loeffelholz, & Bauer, 2005). To better discriminate between forests and timber plantations as well as between the bareland and urban classes, post-classification processing was applied using ancillary data (Appendix A). The MSS classification was re-sampled to a 30×30 m pixel size to allow multi-temporal comparison with the rest of the series. Pre-processing and classification of remote sensing data were performed with PCI 7.0 (2000). Post-classification procedures were performed with ArcMap 9.2 (ESRI, 2006).

Analysis of land cover change

The extent of the original satellite images varied slightly and there were areas of shadows in the 1975 scenes and clouds in the 1985 scenes. Therefore areas without data were subtracted from the whole time series before comparisons, and change calculations were made. To account for these differences, a mask was created containing all pixels with no data from any of the four classifications. This mask was applied to the entire set of time series.

To calculate the extent of each land cover class, we analysed classified maps using ArcGIS 9.2 (ESRI, 2006) and its extension Spatial Analyst. A cross-tabulation procedure between the classifications was processed with IDRISI Andes (Clark Labs, 2006); area change, gains, losses and persistence were calculated as proposed by Pontius, Shusasand, and McEachern (2004). Analysis of change via cross-tabulation is a statistical method to identify signals of systematic processes within a land change pattern (Pontius et al., 2004). Systematic transitions among classes were calculated and examined through the off-diagonal entries of the cross-tabulation matrix. Analysis and mapping of the spatial distribution of transitions, persistence, gains and losses were elaborated with the IDRISI Extension Land Change Modeler (Clark Labs, 2006). To create a map of persistence and changes for the study area, binary change/no change maps were processed for each period. The three resulting maps were added to obtain a map of persistence and change occurrence (1, 2 or 3) for the whole study period.

The annual rate of change for each class was calculated with the formula proposed by Puyravaud (2003):

$$r = (1/(t_2 - t_1)) \times \ln(A_2/A_1),$$

where A_2 and A_1 are the class areas at the end and the beginning, respectively, of the period being evaluated, and t is the number of years spanning that period.

Accuracy assessment

Accuracy assessment involves identifying a set of sample locations (ground verification points) that are visited in the field. The land cover found in the field is then compared to that mapped in the image for the same location by means of confusion matrices. Validation of the 1999 ETM+ and 2008 TM land cover maps was accomplished using 280 independent ground control points. For plantations, only points from stands older than 12 years were considered. The 1975 MSS and 1985 TM land cover maps were verified based on interpretation of the ground control points that had not changed over time (219 and 255 points, respectively) using expert knowledge. For the 1975 MSS land cover maps, an additional map of plantations dated 1970 (INFOR, 1970) was georeferenced and used to identify control points for this class. Classification accuracy was first validated after maximum likelihood classification and then again after post-classification modifications using cartographic information in GIS. Overall accuracy and Cohen's Kappa Index of Agreement (KIA) were calculated for each classification (Lu et al., 2004; Shao & Wu, 2008). Confusion matrices were processed using the Arc View Extension Kappa Tools 2.1a (Jenness & Wynne, 2006).

Results

Area change and change rates of land cover types

Over the whole study period, shrubland was the predominant land cover type, although it declined at an annual rate of −0.7%, from 43.3% of the study area in 1975 to 33.9% in 2008 (Fig. 2). Forest showed the largest decline in relation to its area, with only about 58% (113,605 ha) of its extent in 1975 (195,773 ha) remaining in 2008, and an annual decline of −1.7%. Pasture declined slightly at an annual rate of −0.2%, with around 94% (169,216 ha) of the 1975 extent (178,232 ha) remaining in 2008.

Other land cover types experienced an overall expansion. Thus, agriculture increased annually by 1.1% and expanded to 144% (265,102 ha) of the area occupied in 1975. Bareland reached 157% of its 1975 extent in 2008, with an annual growth rate of 1.4%. A large increase was detected for urban areas as well, with an annual growth rate of 2.7%; these areas occupied 5.6% of the study area in 2008, 241% of its 1975 extent. Timber plantation cover increased by over 288% in 2008 compared to the area in 1975; although the annual change rate was the highest among all classes (3.2%) for the 1999–2008 period, timber plantations covered only 3.4% of the study area in 2008 (Fig. 2).

Land cover changes did not occur at equal rates during all time intervals (Fig. 3). Between 1975 and 1985, forest experienced a strong loss at an annual rate of −3.7%. This annual rate declined to −0.3% and −1.5% for the 1985–1999 and 1999–2008 periods, respectively. Overall, forest losses during the three study periods were offset by about one third by forest gains. During the period of highest forest loss (1975–1985), shrubland cover increased at an annual rate of 0.2%. From 1985 to 1999, the amount of shrubland decreased at an annual rate of −0.6%, reaching a maximum annual loss of 2.0% during the 1999–2008 period. Overall shrubland losses during the three study periods were offset by shrubland gains by about two thirds. Nonetheless, half of these offsets came from forest to shrubland conversion and should therefore not be considered vegetation gain.

Agriculture rose very slightly between 1975 and 1985, but expanded at annual rates of 1.7% and 1.2% during the 1985–1999 and 1999–2008 periods, respectively. Urban areas spread at an annual rate of 5.7% between 1975 and 1985 and continued expanding at annual rates of 1.0% and 2.0% during the 1985–1999 and 1999–2008 periods, respectively. Bareland increased at annual rates of 3.3% and 4.0% during the 1975–1985 and 1999–2008 periods, respectively, but decreased at an annual rate of −1.7% during the 1985–1999 period. Gains and losses in pasture cover were high but compensated for each other over the whole study period, resulting in annual rates of −1.3%, 1.1%, and −0.9% during the periods 1975–1985, 1985–1999 and 1999–2008, respectively. The area of timber plantations remained relatively stable between 1975 and 1999, but experienced an important expansion between 1999 and 2008, with an annual rate of 10.6%.

Spatial distribution of changes

The spatial distribution of intensity and patterns of land cover changes and persistence is shown in Fig. 4. During the entire study period, we found that 28.2% of the study area was subject to only one change, 31.7% was subject to two changes, 16.8% changed in all three time periods, and only 23.3% of the pixels remained unchanged. The majority of the unchanged pixels (1975–2008) were shrubland (42.5%) and agriculture (25.3%), followed by forest (11.1%), urban (8.2%) and pasture (6.6%). An extent of 2.5% of the study area was identified as permanent bareland.

The most intense change dynamics were located in the coastal zone, where frequent exchanges between pasture and shrubland, as well as between pasture, bareland and agricultural areas (particularly rotations between pastures, herbaceous crops and fallow cycles), were found. Timber plantations, generally located in the flat coastal zone, increased and showed relatively high spatial variability due to rotations between plantations and logged areas at the north-west coast and further south on the coastal plains. In mountainous areas, changes were less frequent and mainly consisted of the conversion of forest to shrubland and, to a lesser extent, shrubland to forest. Agriculture expanded across the entire study area, particularly in the flat valleys from the coast to around Santiago. Although the bottoms of some valleys remained as agriculture throughout the 33 years, an increase in agriculture occurred in the foothills at the expense of shrubland and pasture. The increase in urban areas was related to the rapid growth of Chile's capital, Santiago, and the urban agglomeration of major cities located in the north-west part of the study area. Expansion of urban areas was characterised by only one change throughout the three time periods and an aggregated spatial pattern.

Change trajectories among land cover types

The most consistent trend of inter-class change between 1975 and 2008 was a progressive loss of natural vegetation cover (Fig. 5). Between 1975 and 1985, the major changes were a conversion of forest to shrubland (50,351 ha) and of shrubland to

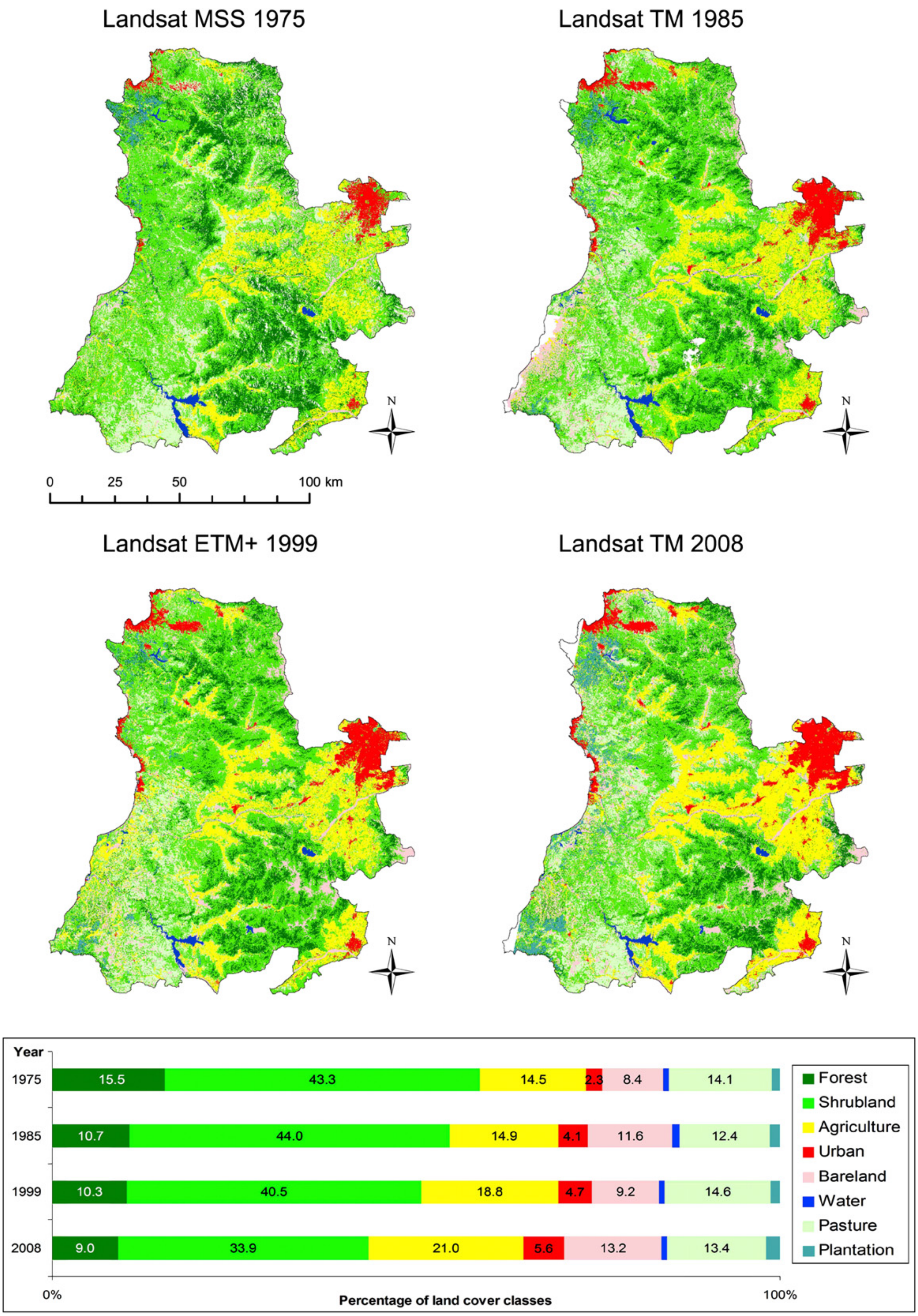
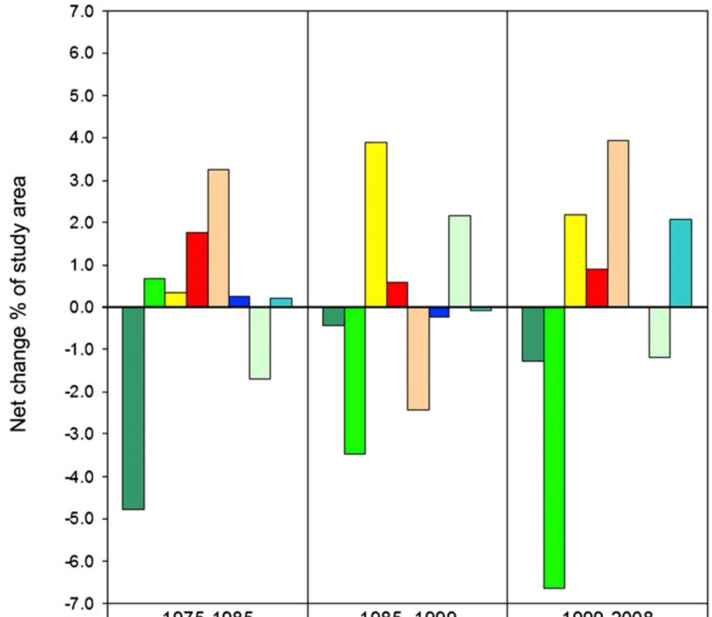


Fig. 2. Land cover maps of the study area in Central Chile for the years 1975, 1985, 1999 and 2008 and comparison of the respective extents of land cover classes by percentage of study area (study area = 1,265,204 ha).



	1975-1985		1985- 1999		1999-2008	
	Gains	Losses	Gains	Losses	Gains	Losses
Forest	4.5	-9.3	4.9	-5.3	4.2	-5.5
Shrubland	19.5	-18.8	15.2	-18.7	12.5	-19.2
Agriculture	6.0	-5.7	8.6	-4.7	7.7	-5.6
Urban	2.0	-0.3	1.1	-0.5	1.3	-0.4
Bareland	9.0	-5.8	5.5	-8.0	9.3	-5.4
Water	0.3	-0.1	0.1	-0.4	0.2	-0.2
Pasture	7.7	-9.4	9.5	-7.3	8.1	-9.3
Plantation	0.9	-0.7	0.9	-1.0	2.7	-0.6

Fig. 3. Net change (i.e. gains minus losses), gains and losses for each land cover class as a percentage of the study area for the periods 1975–1985, 1985–1999 and 1999–2008 and for the whole study period, 1975–2008.

bareland (21,689 ha), pasture (14,022 ha) and urban areas (9356 ha, Fig. 5). Between 1985 and 1999, the major contributions to net change were the conversion of shrubland to agriculture (40,338 ha) and pasture (18,726 ha). Shrubland regained some area from bareland (14,647 ha), whereas bareland contributed to a gain in agriculture (12,589 ha). Between 1999 and 2008, the foremost change was a loss of shrubland that contributed to increases in bareland (42,132 ha), agriculture (19,356 ha) and timber plantations (14,973 ha). In contrast to previous periods, shrubland regained a small area from forests (9968 ha, Fig. 5).

Accuracy assessment

Classification accuracy increased notably after applying post-classification procedures, from overall agreements of 56.2%, 60.8%, 60.9%, and 72.3% to 65.8%, 77.3%, 78.9%, and 89.8% for the 1975 MSS, 1985 TM, 1999 ETM+, and 2008 TM images, respectively (results not shown). Cohen's Kappa Index of Agreement (KIA) was 63.4%, 73.8%, 75.8% and 88.3% for the set of post-processed images.

Discussion

Patterns of landscape change

Human interactions with ecosystems are inherently dynamic and complex, and any categorisation of these is an oversimplification. However, there is little hope of understanding these interactions without such simplifications (Ellis & Ramankutty, 2008). Working at the broad scale of this study has the advantage of providing general trends at the regional scale that are useful for landscape planning and serve as a basis for analyzing drivers of land cover change. However, it has the disadvantage of lower detectability of pattern and processes at the scale of land-use units in the real world (e.g. a field). Nevertheless, informal interviews that were conducted accompanying the field survey give an important complementary source of information to interpret detected changes at this regional scale and reinforce that observed changes followed similar ground level land-use patterns throughout the study area.

Our analysis of land cover change in Mediterranean Central Chile reveals a general trend of a continuous reduction in natural vegetation, i.e. forest and shrubland cover, that in turn has led to an increase in provisioning ecosystem services such

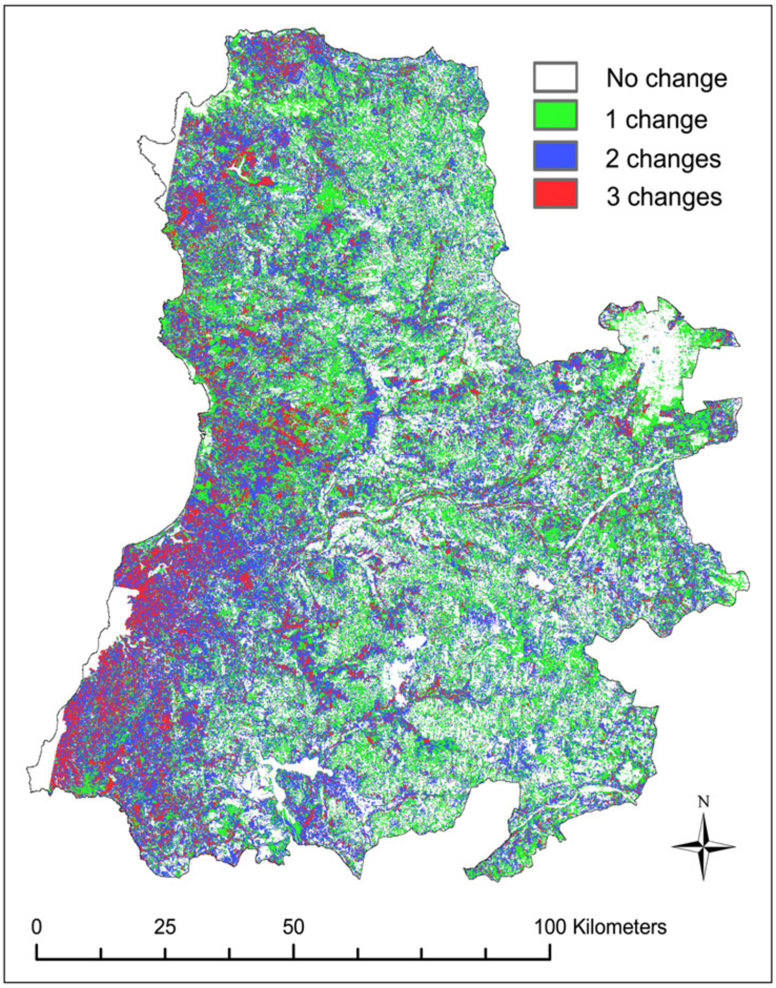


Fig. 4. Distribution of persistent pixels (i.e. those that never changed land cover type) and of pixels that showed one, two or three changes across the three periods analysed from 1975 to 2008.

as food and timber production. This process takes place as a progressive modification from forest to shrubland vegetation, and a highly dynamic conversion between shrubland and human-induced types of land cover. Nevertheless, deforestation rates in this region are relatively low compared to rates in temperate forests in south-central Chile (Echeverria et al., 2006). This is probably due to the fact that Central Chile has been densely populated since the times of colonisation and major conversions of forest cover had taken place long before the 1970s (Conacher & Sala, 1998).

However, a relatively high amount of shrubland, the predominant vegetation cover in this semiarid landscape, was lost as a consequence of conversions to intensive land uses, chiefly expanding agriculture and timber plantations and, to a lesser extent, urbanization. This can be explained by an increase in local demand due to population growth and an open market policy initiated after Chile's economic crisis at the beginning of the 1970s (Camus, 2000; Silva, 2004). Agriculture and forestry thereafter became the most important competitive producers (Camus, 2000). The strong increase in agriculture has been stimulated by a combination of market liberalisation, incentives for new export-oriented crops, introduction of new irrigation technologies, and improvements in road infrastructure (Valdés & Foster, 2005). The expansion of timber plantations was mostly a result of a government subsidy for tree-planting initiated in 1974 (Decree 701), which stimulated the planting of *P. radiata* and *E. globulus* (Aronson et al., 1998). In the case of Central Chile, the rate of increase of timber plantations was the highest of all classes in the 1999–2008 period. However, the expansion of timber plantations did not result in major conversions of forest, as it did in southern Chile (Echeverria et al., 2006). Rapid expansion of urban areas, chiefly in the 1975–1985 period, coincided with the abolishment of the urban limits by the Ministry of Housing and Urbanism (Decree 420) in 1979 and the liberalisation of the urban land market, both until 1985 (Kusnetzoff, 1987).

In our study region, forest loss patterns consisted mainly of the conversion of forest to shrubland and the reduction of forest to remnants located on steep hills, where intensive use by humans is constrained by topography. The transformation of forest to shrubland has been described as a continuous degradation of sclerophyllous forest, mostly driven by permanent grazing pressure, firewood collection and charcoal production (Armesto et al., 2007; Balduzzi et al., 1982; Fuentes, Hoffmann, Poiani, & Alliende, 1986; Rundel, 1999). In addition, successional recovery of forest is largely constrained by water availability, soil erosion, lack of seed banks, disturbance by human-induced fires and limited regeneration capacities of forest species as

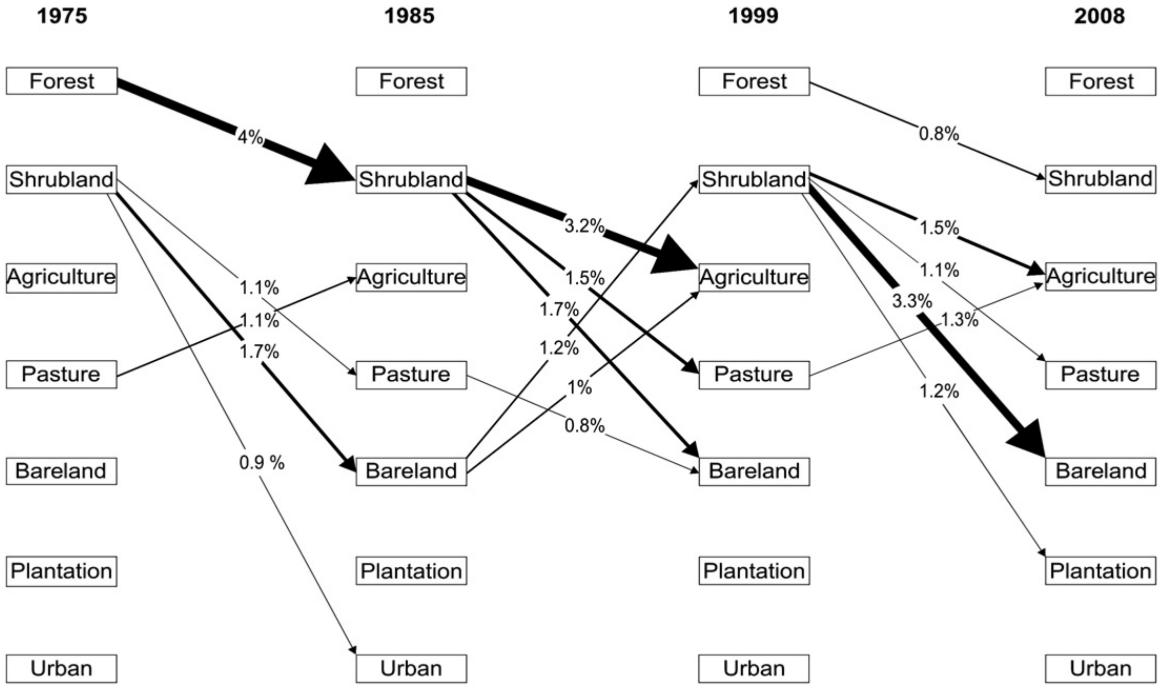


Fig. 5. Major change trajectories and their contributions to net change in percentage of the study area (thick lines correspond to net change >3.2%, intermediate lines correspond to net changes between 1.6% and 3.2%, and thin lines correspond to net change <1.6%; only net contributions to change >10,000 ha or 0.8% of the study area are represented).

compared to shrubland species (Armesto et al., 2007; Balduzzi et al., 1982; Conacher & Sala, 1998; Fuentes et al., 1986; Jimenez & Armesto, 1992; Montenegro, Ginocchio, Segura, Keely, & Gómez, 2004; Rundel, 1999). This is evidenced in our analysis by the large proportion of shrubland that remained unchanged over the whole study period. It has been argued that the loss of forest and the change toward the predominance of shrubland cover represent a shift to an alternative ecosystem state (Holmgren, 2002). In contrast to deforestation patterns in other dry forest regions, no direct conversion from forest to agriculture (Izquierdo & Grau, 2009) or *vice versa* (e.g. Lasanta-Martínez, Vicente-Serrano, & Cuadrat-Prats, 2005) occurred in the study area.

We detected forest recovery through succession (on about 2.7% of the study area) over the whole study period as other authors have documented in other Mediterranean areas (Serra, Pons, & Saurí, 2008). Nonetheless, the latter process has been halted, as mentioned above, by different physical and ecological factors. Thus, shrubland acts as a highly dynamic compartment with large gains and losses over the three study periods as compared to other land cover changes. Bidirectional changes (i.e. gains and losses) in shrubland cover resulted in the following spatial patterns: (1) exchanges between shrubland and agriculture, agriculture and pasture, and pasture and shrubland took place in small patches scattered throughout the relatively flat areas and partly explain the high dynamics detected in the studied landscapes; (2) such exchanges resulted in a net loss of shrubland due to land-use intensification and agglomeration; and (3) from 1999 onward, these patterns have spread to hillslopes, as indicated by the appearance of relatively large continuous patches of newly cleared bareland and agriculture on the lower hillslopes, which were formerly covered by shrublands. This was motivated by important governmental subsidies to encourage irrigation schemes (Maletta, 2001) and a collapse of the capacity of the flat areas to sustain the increasing expansion of agricultural land.

Consequences of land cover change: implications for landscape planning and management

Vegetation loss and degradation reduce precipitation infiltration and runoff regulation, which promotes soil erosion and has a negative impact on ground water recharge (Conacher & Sala, 1998; Millennium Ecosystem Assessment, 2005). In addition, vegetation cover is highly correlated with water balance and regional climate regulation (Feddema et al., 2005; Foley et al., 2005; Pielke, 2005). Changes in land use by humans and the resulting alterations in surface features and biogeophysical processes influence weather and climate more immediately than the carbon cycle (Bonan, 2008; Pielke, 2005). Land-use decisions therefore have consequences for the structure and function of ecosystems and affect environmental goods and services; these decisions also affect humans in ways that go beyond the immediate land-use situation (Turner et al., 2007). The continuous degradation of the vegetation cover could have a strong impact on human livelihood and well-being in Central Chile as well as other dryland landscapes, as there are increasing water demands for agriculture (Cai, Ringler, & You, 2008) and human consumption due to large population increases.

Environmental problems like degradation, loss of biodiversity and decreases in productivity accumulate over the long term and have non-linear effects on regional to global scales (DeFries, Foley, & Asner, 2004; Foley et al., 2005). Consequently, strategies for adapted land use, including the optimisation of the spatial configuration of uses and restoration of the natural vegetation cover should be developed quickly. Vegetation cover within the landscape mosaic must be carefully considered in planning to sustain habitat and regulation functions and enhance the productive capacities of the landscape. Strategies should go beyond preservation within protected areas and logging restrictions along rivers and streams (Turner et al., 2007). For instance, Rey Benayas, Bullock, and Newton (2008) proposed the "woodland-islet in agricultural seas" model to conciliate agricultural production and conservation or restoration of native woodlands. Closer monitoring is needed for cattle grazing stocks to establish guidelines for an adapted carrying capacity, as cattle graze on pastures, shrubland and in forests, which are all mainly private land and do not have adequate use restrictions. The repercussions of firewood extraction and charcoal production have hardly been quantified in Central Chile, but we know that firewood and derived charcoal provided around 18% of the national energy supply between 1990 and 2007, while firewood consumption doubled in this period (CNE, 2008). Some estimates by Dubroeucq and Livenais (2004) in northern Chile show that the impact of firewood extraction on vegetation cover should not be underestimated.

Apart from the need for land-use planning, restoration and rehabilitation are important issues in drylands (Le Houerou, 2000; Vallejo, Aronson, Pausas, & Cortina, 2006). Holmgren and Scheffer (2001) postulated that there might be a window of opportunity for passive restoration through the exclusion of herbivores in ENSO years due to higher water availability, which Gutiérrez, Holmgren, Manrique, and Squeo (2007) have experimentally shown for drier zones further north in Chile. It could be especially interesting to use this strategy to establish buffer zones and corridors between remaining old growth forest, which were detected in this study as stable forest areas. Land-use planning to ensure the long-term maintenance of landscape functions that are of common societal concern has not yet been established in Chile, where territorial planning legislation mainly focuses on urban areas, infrastructure and industrial development. Recently, some efforts have been made in landscape planning for protected areas (Oltremari & Thelen, 2003). Also, forms of adaptive and multifunctional land use like mixed agroforestry systems should be encouraged as an alternative to monoculture cropping and crop pasture rotations (Aronson et al., 1998; Ovalle, Del Pozo, Casado, Acosta, & de Miguel, 2006).

Conclusion

Several uncertainty factors underlie the classification of satellite imagery into land cover types, and such classification is never completely accurate (Shao & Wu, 2008). However, this work has estimated the extent of land cover in Mediterranean Central Chile, characterised the respective changes, and assessed the dynamics and stepwise vegetation cover loss that has taken place over the last 33 years. Our case study provides further evidence of how Mediterranean regions show a constant transformation of their ecological systems. This analysis illustrates how natural vegetation cover tends to diminish in a very subtle and slow fashion due to passive revegetation that partly counterbalances vegetation loss. Nevertheless, forest cover is being degraded, and many areas do not recover, but remain as shrubland. Shrubland, in turn, is lost to intensive land uses like agriculture and timber plantations. Land-use changes in Mediterranean regions must not always be interpreted as the loss of a specific set of ecological conditions based on values established in the world of the West, but as a change in the ecosystem services that dynamic land cover types provide to humans in culturally distinct regions. However, our interviews conducted during the field survey revealed a high awareness of local population regarding the benefits that forests provide and the consequences of forest loss, including reduced water provision and erosion problems. The successful identification of change trajectories provides a critical component for land use, conservation and restoration planning in dry landscapes. The investigation of regional dynamics provides a basis for future analysis of drivers and circumstances that enhance change or stability of land cover.

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Appendix. Supplementary material

Supplementary data associated with this article can be found in the online version, at doi:10.1016/j.apgeog.2009.12.003.

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