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Abstract: Due to the high sensitivity of mountain landscapes to environmental changes, the study of land cover dynamics has become an essential tool for guiding management policies. Since the second half of the twentieth century, the Cantabrian Mountains (NW Spain) have been substantially altered by the loss of traditional management practices and, more recently, by the new environmental schemes developed by the Regional Government. This area is a biodiversity hotspot, representing the south-western-most distribution limit for a large number of species in Europe. Therefore, small changes in landscape patterns can result in biodiversity losses. In this study, we analyzed land cover changes in the Cantabrian Mountains from 1991 to 2004 by means of remote sensing techniques, identifying the main driving forces and classifying the territory according to its risk of land cover change. Forest expansion and loss of shrublands were the two major trajectories of change apparent during this period. When modeling the occurrence of these land cover changes, we found that performance of models was related to the nature of the change. The most accurate models were associated with processes of secondary succession, i.e. forest expansion (78.6%), while the least accurate models related to changes linked with management decisions, i.e. loss of shrubs (61.8%). The main drivers of change were variations in the number of goats (for the forest expansion model) and changes in the number of head of sheep and cattle (for the loss of shrubs model). Topographic conditions (altitude and slope) were relevant in both models. Our approach proposes an explicit decision support tool for landscape managers, allowing better identification of the areas where they should focus their attention.

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Using predictive models as a spatially explicit support tool for managing cultural landscapes.

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Using predictive models as a spatially explicit support tool for managing cultural landscapes

ABSTRACT

Due to the high sensitivity of mountain landscapes to environmental changes, the study of land cover dynamics has become an essential tool for guiding management policies. Since the second half of the twentieth century, the Cantabrian Mountains (NW Spain) have been substantially altered by the loss of traditional management practices and, more recently, by the new environmental schemes developed by the Regional Government. This area is a biodiversity hotspot, representing the south-western-most distribution limit for a large number of species in Europe. Therefore, small changes in landscape patterns can result in biodiversity losses. In this study, we analyzed land cover changes in the Cantabrian Mountains from 1991 to 2004 by means of remote sensing techniques, identifying the main driving forces and classifying the territory according to its risk of land cover change. Forest expansion and loss of shrublands were the two major trajectories of change apparent during this period. When modeling the occurrence of these land cover changes, we found that performance of models was related to the nature of the change. The most accurate models were associated with processes of secondary succession, i.e. forest expansion (78.6%), while the least accurate models related to changes linked with management decisions, i.e. loss of shrubs (61.8%). The main drivers of change were variations in the number of goats (for the forest expansion model) and changes in the number of head of sheep and cattle (for the loss of shrubs model). Topographic conditions (altitude and slope) were relevant in both models. Our approach proposes an explicit decision support tool for landscape managers, allowing better identification of the areas where they should focus their attention.

KEY WORDS: Rural abandonment; LANDSAT imagery; Socio-economic drivers; predictive modeling.

1. INTRODUCTION

Land cover and land use changes (LCLU) have been widely studied worldwide (Reid et al., 2000; Achard et al., 2006; Seabrook et al., 2007; Izquierdo and Ricardo, 2009; Townsend et al., 2009). Understanding the relationships between land cover patterns and their associated processes is extremely important for scientists, landscape managers and policy makers designing nature conservation strategies aimed at preserving some of the unique characteristics of landscapes (Kates et al., 2001). In the European context, there is increasing concern about detecting landscape changes and their ecological consequences in mountain systems (MacDonald et al., 2000; Mottet et al., 2006; Gellrich et al., 2007), since these systems provide important ecosystem services (cultural, provisioning, regulating and supporting) contributing to both highland and lowland economies. Mountain systems constitute important biodiversity hotspots, since their biota is adapted to specific, narrow environmental limits, which makes them extremely sensitive to slight perturbations (Millennium Ecosystem Assessment, 2003). Therefore, early detection of changes in land cover patterns has become an important target in these ecological systems since it allows identification of the main landscape drivers and, consequently, the design of realistic management policies.

Through the centuries, human activity has played a decisive role in managing and shaping the landscape of Mediterranean mountain systems (Grove and Rackham, 2000). Studies related to land cover trends in these systems have focused mainly on changes resulting from agricultural abandonment. These are widespread, having begun in the early 20th century (Rey-Benayas et al., 2007). In the Iberian Peninsula, additional pressures result from declines in the transhumance system (Molinillo et al., 1997; Gómez and Lorente, 2004; Vicente-Serrano et al., 2005), which have been especially noteworthy in the Cantabrian Mountains range. These have involved a gradual loss of traditional management (based on grazing, cutting and burning; Calvo et al., 2007), favouring a process of landscape homogenisation that impacts biodiversity and threatens cultural landscape heritage (Jongman, 2002).

Remote sensing techniques have become essential tools for land cover studies at regional scales because of their capacity to provide a temporal series of information on the terrestrial land surfaces and, thus, give an integrated understanding of changing processes in the landscape (see for example Michalski et al., 2008). Predictive models, such as those based on regression, may provide a framework for identifying the driving forces behind land cover changes (Serra et al., 2008). The occurrence of land cover change can be successfully predicted across space by combining statistical models with spatially explicit data in a geographical information system (GIS) environment (Schneider and Pontius Jr., 2001). This combination of techniques and data represents a powerful tool for defining priority areas in terms of land management and biodiversity conservation (Lehmann et al., 2002).

Within the framework of the European Union (EU) and the Natura 2000 Network requirements (Habitats Directive, 92/43/EEC; Annon, 1992), such models can provide important input to the design and application of integrated management strategies. Land use planning in mountain areas, which has traditionally focused on local actions and their short-term consequences, could then be replaced by unified policies addressing the preservation of the natural spaces network as a whole. Furthermore, maps illustrating predicted changes over time could empower land users to understand long-term outcomes of decisions (see for example, Binkley and Duncan, 2009), helping to connect people and ecology.

In this study, we evaluated land cover changes in the Cantabrian Mountains during the period 1991 to 2004 using LANDSAT images. We then developed spatially explicit models of land cover change in order to predict where future transitions may occur in the landscape, as well as to define the main forces (biophysical and human) driving the observed changes. The results provide a useful support tool for landscape managers, since it allows the identification of priority areas in terms of conservation.

2. METHODS

2.1 Study area

The study area is located at the southern slope of the Cantabrian Mountains in the León province (NW Spain) (Figure 1). It includes 28 municipalities, covering 3,266 km^2. Altitude ranges from 877 to 2,412 m.a.s.l. This area lies on the boundary between the Atlantic/Eurosiberian and Mediterranean biogeographical regions, where four bioclimatic belts ranging from supramediterranean to subalpine can be identified (Rivas-Martinez et al., 1987). Average annual rainfall varies from 700 to 1,573 mm. Mean annual temperature varies from 8.2º C to 9.9º C. Atlantic vegetation is generally located on shaded and humid northern slopes. It mainly consists of deciduous forests (*Fagus sylvatica*, *Betula pubescens*, *Quercus petraea*, *Q. robur*) and heathlands of *Calluna vulgaris*. Mediterranean vegetation (*Quercus pyrenaica*, *Q. ilex*) mostly occurs on sunny and drier slopes. The lithology is also diverse, from massive Carboniferous limestones to slate with coal, sandstones, crystal quarzites and conglomerates (Gómez and Rodríguez, 1992). Among the driving forces that have shaped the landscape of this area over the centuries, transhumance has played an important role. It consists of the movement of flocks of thousands of sheep and goats from the south-Iberian “dehesa” systems to the Cantabrian pastures above the treeline during the summer. This activity has provided an important source of income for most of the municipalities of this area for centuries (Gómez and Rodríguez, 1992). Due to the complex orography, agricultural activities are concentrated on the slopes close to the villages and in valley bottoms. Timber has traditionally been extracted from Atlantic forests for a range of purposes. Since the beginning of the twentieth century, the decline in the transhumance system, the abandonment of agricultural areas due to their lower profitability and the process of population emigration from the countryside to the cities (especially marked since 1960), have resulted in the loss of traditional management practices across the territory. Furthermore, other activities, such as the opening of coal and opencast talc mines, construction of large reservoirs and afforestation of thousands of hectares with conifers, have transformed the landscape.

2.2 Data sources

Land cover was mapped using a temporal series of LANDSAT images for the years 1991, 1995, 2000 and 2004 (Table 1). We selected images acquired during the summer season in order to obtain the lowest cloud and snow cover (the latter being present sometimes even in late spring).

LANDSAT images were geometrically corrected following the polynomial method proposed by Palá and Pons (1995), which uses ground control points extracted from aerial photographs as references and a 30 m resolution Digital Elevation Model (DEM) to minimize geometric errors. It was implemented using Miramon software (Pons, 2002). Around 60 ground-control points per image were used for geometric correction. Two thirds of these were used to correct the images; the remaining points were used to validate the results. The Root Mean Square Error (RMSE) was lower than the pixel size in all images (RMSE±SD=16.16± 4.78). Radiometric correction was based on the algorithms developed by Markham and Barker (1986) and Morán et al. (1992), while atmospheric correction followed the transmittance model (COST) proposed by Chavez (1996). Down-welling transmittance values for bands five and seven were taken from Gilabert et al. (1994), since atmospheric conditions in their study area were more similar to ours than the area used by Chavez (1996). In order to compensate for differential solar illumination due to the shape of the terrain, a topographic correction was applied following the non-Lambertian C-Correction method (Teillet et al., 1982; Riaño et al., 2003). As each individual image was classified independently, the image series was not normalised.

Other data sources used in this study include a DEM at a 30-meter resolution and several vector layers showing urban areas, water surfaces and coniferous afforestation (see Table 2). Since these vector layers are inaccurate because of their coarse scale (1:200,000), they were edited and corrected on screen using high resolution orthophotographies. Socio-economic data accounting for changes in livestock density (number of heads of sheeps, cows and goats) and human population density were recorded at the municipality level. All vectorial data were rasterized and resampled to match the LANDSAT 30-m spatial resolution.

2.3 LANDSAT image classification

A supervised classification (Maximum Likelihood method; Conese and Maselli, 1992; Martin et al., 1998; Shalaby and Tateishi, 2007; Schulz et al., 2010) was applied to produce a land cover map per year of study (1991, 1995, 2000, 2004). The variables involved in the process (see Table 2) were: (i) LANDSAT bands from 1 to 7 (excluding band 6 because of its coarse resolution). (ii) Two vegetation indices derived from LANDSAT images: the Normalized Differenced Vegetation Index (NDVI) (Rouse et al., 1973) and the standard Greenness Index (GI) of Tasselled Cap Transformation (Kauth and Thomas, 1976), as a measure of total photosynthesis and vegetation productivity. (iii) A spatial enhancement variable (texture), calculated as the average of the variance from the third band of each image measured within a mobile window 3x3 pixels. (iv) Topographic variables (altitude and slope) derived from the DEM.

Initially, we defined nine land cover categories: water, rock/bare ground, meadows, climatic pasturelands, heathlands, shrublands dominated by *Genista* spp., coniferous afforestations, forests (including deciduous and evergreen formations) and urban areas. After several preliminary classifications, we identified various conflicting assignations associated with the spatial complexity and heterogeneity of the landscape mosaic in the study area. Particularly relevant were problems of misclassification when dealing with the category “coniferous afforestations”, since it includes a wide range of development states, from mature to recently planted patches showing confusion with other land clover class as shrublands. This uncertainty was especially relevant in recently disturbed areas (young plantations). For this reason, coniferous afforestations, together with water and urban areas, were masked out and removed from the classification process. The remaining categories were grouped into four final classes: rock/ bare ground, herbaceous vegetation (including meadows and pasturelands), shrublands (including shrublands and heathlands) and forests.

High-resolution orthophotography (years 2000 and 2004) and fieldwork data (2005) were used to define at least 20 training polygons per category with a size of 50 pixels (4.5 ha), which were distributed throughout the study area following a stratified sampling design. Using the same data sources, as well as Spanish Forest Inventories, we created a set of validation layers containing 100 points per category and year.

Classification accuracy was assessed using a confusion matrix, which allowed validation points to be compared with the classified land covers. In order to interpret the matrix, we evaluated persistence and “swaps” (transitions between gains and losses in categories), according to Pontius Jr et al. (2004). Since coniferous plantations, water and urban areas were considered as constant land covers during the study period, we only assessed dynamics within rock/bare ground, herbaceous vegetation, shrubland and forest.

2.4 Modeling land cover changes

Principal Components Analysis (PCA) has been widely used in pre-classification change-detection approaches (see examples in Lu et al., 2004; Cakir et al., 2006; Pu et al., 2008; Deng et al., 2008), but less often in post-classification analyses. We applied a PCA on the temporal series of classified LANDSAT images (1991, 1995, 2000 and 2004) to enhance the detection of regions of change under the following assumption: We assigned values of 1 to 4 to the land cover categories, considering them as continuous, incremental stages of a secondary succession process: rock/bare ground (1), herbaceous vegetation (2), shrubland (3) and forest (4). Therefore, higher values will always mean greater complexity of community structure and ecosystem functioning. Consequently, landscape trends involving evolution towards more complex ecosystem structures would imply a shift from 1, 2, or 3 towards 4. On the other hand, land cover changes from 4, 3 or 2 towards 1 would imply a loss in complexity of ecosystem structure backwards in the ecological succession due to perturbations. Subsequently, an unsupervised classification (ISODATA) was carried out on the first four principal components obtained from the previous step. The resulting cluster classes, corresponding to different categories of land cover change, were used in further analyses. For any given cluster class, a pixel in the landscape will be classified as “1” (present), belonging to their class, or “0” (absent) not belonging.

A sample of 20,000 systematic sample points was then generated throughout the study area to record different trajectories of land cover change (i.e. cluster classes). The sampling size was proportional to the surface corresponding to each change. Topographic and socio-economic data (see Table 2) were also assigned to those locations and recorded in a database structured around the cluster category. For each of these trajectories, we selected a prevalent data sample within each presence/absence class but with equal sample numbers in each class. We then applied binary logistic regression (BLR) to model the probability of the occurrence of the main land cover changes as a function of the independent variables.

Before running the regression analysis, we checked for high Spearman correlations among the topographic and socio-economical variables, which would hinder the stepwise selection procedure. Finally, we assessed the significance of each of the independent variables, as a function of “change” or “no change”, using Mann–Whitney *U*-tests. Only significant variables ($p < 0.05$) were retained in the final modeling analysis.

The selection of variables that best fitted into our models was made by means of a backwards stepwise regression. The analysis begins with a full model, where independent variables are removed in an iterative process (Hosmer and Lemeshow, 1989) based on a Wald algorithm. The ability of the model´s predictions to discriminate between the response classes was evaluated using Relative Operating Characteristics (ROC) (see Pontius Jr and Schneider, 2001; Braimoh and Vlek, 2005; Mathew et al., 2009). ROC values range from 0.5 (for a model that assigns the probability at random) to 1 (for a model that perfectly assigns the probability of observing the trajectory in the landscape). The logit function of the probability obtained from the models was converted, through an inverse logistic transformation, into a map showing the probability of change occurrence on a scale of 0 to 1.

GIS analyses were done using ERDAS® IMAGINE 8.5 and IDRISI KILIMANJARO 14.1 and statistic analyses were performed with SPSS 16.0.

3. RESULTS

3.1 Classification results and accuracy assessment

Table 3 shows LANDSAT classification accuracy for the complete temporal series. In all cases, the average accuracy was greater than 85%, which indicates a good level of reliability. The lowest producer’s accuracy level corresponded with herbaceous vegetation, which was under-classified (values ranging from 73.91 to 85.22%); while the highest value was for rock/bare ground (values ranging from 87.13 to 97.09 %). The lowest user’s accuracy corresponded to shrublands, which were over-classified; while the highest value was for the rock/bare ground category.

3.2 Land cover change monitoring

Fig. 2 and Table 3 show that, at the beginning of the 1990s, the landscape in the study area consisted of a shrubland matrix (42.99% of the total area), with patches of different land cover. Forest patches represented 13.99% of the area and patches directly linked with human activities occupied respectively the 21.46 % (grasslands), 0.77% (water reservoirs) and 1.12% (urban areas). Rock and bare ground occupied 11.91% and coniferous plantations covered 7.77% of the study area.

From 1991 to 2000 the area occupied by shrublands only increased from 42.99% to 43.09% (less than one percent), while rock/bare ground and herbaceous vegetation showed an increase in area by 3.28 % and 5.41 %, respectively. These increases corresponded with forests reduction from 13.99% to 12.34% cover. The most remarkable changes in these vegetation categories occurred during the period 2000-2004, when the cover of both herbaceous vegetation and shrublands decreased (5.31% and 5.93% respectively), while the area of rock/bare ground and forest increased by 13.69% and 16.63%, respectively. For the study period as a whole, shrubland cover decreased a 5.72%, while both rock/bare ground (17.69%) and forests (2.88%) increased. Note that these percentages of change were calculated based on the surface occupied by each land cover in 1991. The area occupied by herbaceous vegetation underwent a slight reduction through the study period.

The cross-tabulation matrix between 1991 and 2004 (Table 4) provides information about systematic transitions between land cover types. For example, herbaceous vegetation, which did not undergo significant net changes (less than 1% decrease), was very dynamic (extensive swapping). In 2004, only 53.26% of herbaceous cover remained invariant from 1991, with 16.90% changing to shrubland, 5.94% to forest and 15.90% to rock/bare ground. Concurrently, 27.06% of the initial shrubland cover, 10.68% forest and 9.00% rock/bare ground shifted to herbaceous vegetation, making the net change of this land cover category almost null.

The 2.88% net change in forest cover was mainly associated with a swap from shrubland to forest (33.50%) and from herbaceous vegetation and rock/bare ground to forest (5.94 % and 1.55 % respectively). During the whole study period, forest mainly change towards shrubland (8.73%) and herbaceous vegetation (10.68%).

3.3 Land cover change modeling

We obtained 12 cluster classes (i.e. possible land cover changes from 1991 to 2004), which were interpreted according to the temporal sequence of land covers detected for the study period. Five cluster classes represented non-change situations, with the remaining seven classes representing different trends of change. Two of these classes of change corresponded with balanced situations between herbaceous-shrublands and shrublands-forests (23.83 and 18.74% of the sample points respectively). A 9.3% of the points represented areas of random change through time, corresponding with two cluster classes. Another cluster category (5.13%) recorded the processes of perturbation involving the loss of vegetation cover between 1991 and 1995, followed by a later recovery. The two remaining cluster classes corresponded with forest expansion and a consistent loss of shrubland throughout the whole timeframe (4.82% and 4.9% of the sample data respectively). Only the latest two classes were considered in the following statistical analyses.

3.3.1 Forest expansion model

Mann–Whitney *U*-tests on the explanatory variables showed that all of the following variables could significantly explain forest expansion changes: altitude ($p< 0.01$), slope ($p< 0.01$), change in population density ($p< 0.01$) and temporal shifts in number of head of cattle ($p< 0.01$), sheep ($p< 0.02$) and goats ($p< 0.01$). On average, the change "increase of forest cover" predominantly occurred at altitudes of around 1,255 m (ranging from 1,241 to 1,267 m) and slopes of 5.09 degrees (ranging from 5.46 to 7.72). Moreover, this change was associated with sites characterized by larger reductions in both population and number of head of sheep than areas where forest did not increase. The Spearman test did not detect strong correlations among variables ($r <0.8$). Therefore, all of the independent variables were included in the binary logistic regression analysis (BLR).

Using backwards stepwise regression, only three variables were retained: altitude, slope and number of head of goats. The estimated regression coefficients are shown in Table 5. Positive values of the standardized logit coefficients indicate that higher values of the independent variables increase the probability of observing the trajectories. The "odds ratio", Exp (β_1), measures the likelihood of observing a trajectory if the independent variable is increased by one unit. When $\beta>0$, $Exp(\beta)>1$, indicating that the odds of observing the trajectory increase, and when $\beta<0$, $Exp(\beta) <1$, meaning the likelihood of observing the trajectory decreases. When $\beta=0$, $Exp(\beta)=1$, and the likelihood of observing the trajectory is not affected. The discrimination of the model, estimated through the area under the ROC curve (AUC), was 0.79. The probability map derived from the logistic function is shown in Figure 3. The highest probability of forest expansion was mainly found in two locations: the whole southern part of the study area, corresponding with the lower peaks of the Cantabrian mountain range in the province of León, and valley bottoms in the northern part of the study area. In the latter, increasing the distance from valley bottoms reduced the probability of observing this change.

3.3.2 Loss of shrub model

Changes in population density ($p= 0.12$) and number of head of goats ($p= 0.17$) were excluded from the subsequent analyses according to the Mann–Whitney *U*-test. The remaining variables were taken into account in successive analyses because they were uncorrelated according to the Spearman test ($r<0.8$): altitude ($p< 0.01$), slope ($p< 0.01$), number of head of cattle ($p< 0.01$) and sheep ($p< 0.01$). Loss of shrubland cover occurred in areas with a mean slope of 8.5 degrees, characterized by significant losses in the number of head of sheep and, at a lower extent, in the number of head of cattle.

The regression model was fitted as a function of the variables altitude, slope and number of head of both cattle and sheep (Table 5). The model discrimination was 0.62, which was lower than the fit of the forest expansion model. The probability map derived from the logistic function is shown in Figure 4. In this case, there were no clear patterns of spatial distribution of the change in the study area.

4. DISCUSSION

4.1 Land cover change monitoring

Land cover changes in the Cantabrian Mountains of León have followed the same trajectories observed in other Mediterranean mountain ranges since the beginning of the 20th century. The loss of traditional management practices (grazing, burning and cutting) have led to abandonment of both pastures and crops, resulting in shrub and forest encroachment (MacDonald et al., 2000; Lasanta-Martínez et al., 2005; Rey-Benayas et al., 2007; Pelorosso et al., 2009; Geri et al., 2010). We observed that, at the beginning of 1990s, almost half of the study area was covered by shrublands as the result of more than one hundred years of decline of the transhumance system (Rodríguez, 2005), increasing depopulation and an ageing population in the countryside (Collantes, 2001). This process continued in the Cantabrian Mountains until 2000, when the trend started to change. The decrease in shrubland cover from 2000 to 2004 can be directly related to new conservation policies adopted in the study area. Four protected areas have been declared since 1990, covering 38% of the Cantabrian Mountains. In these protected areas, specific silvicultural measures have been introduced since 2000 (Gil and Torre, 2007) to favour the regeneration of pastures to support extensive grazing and, at the same time, to create firebreaks to control both the risk and extent of fire. Such measures (particularly shrubland clearance to increase pastureland) have been successfully applied in other regions of Spain (Andalucía, Asturias, Galicia and La Rioja), as well as in other Mediterranean areas (Lasanta et al., 2009). Even in the few years since their introduction, these management strategies have had noticeable effects in the Cantabrian Mountains at regional scale. These include a reduction in arson fires in the study area (Head of Forest Fire Prevention, personal communication). The decrease in the area affected by forest fires can partly explain the 2.88% increase in forest cover recorded throughout the study period. Another key factor was the establishment of restrictive policies in the protected areas, involving a reduction in the incidence of some traditional management practices, such as burning. This is expected to drive secondary succession towards forest stages, as experienced in other protected areas in Spain (Peñuelas and Boada, 2003).

At this point, it is necessary to consider the interpretation of change from satellite images when comparing data from different months. Variations in the spectral behavior of the vegetation, due to its leaf water content, may lead to a certain level of confusion between land cover types, particularly in boundary areas, where the uncertainty is higher (Roy, 2000). Therefore, pixels on the border between shrublands and forests may be classified as a different land cover type depending on leaf moisture content. This effect has been detected in the study area, related to the fact that in 2000, the registered rainfall was 1655.1 mm, which represents less than one half of the value recorded in 2004 (3817.4 mm) (Morán-Ordóñez, unpublished data). Even considering these sources of error, the trend of increase in forest cover (linked with abandonment) compares well with results reported over the last fifty years in other mountain ranges of northern Spain, including the Pyrenees (Roura-Pascual et al., 2005), as well as other areas of the Cantabrian Mountains (Rescia et al., 2008). Moreover, similar results were obtained by the Third National Forest Inventory 1992-2003 (Ministry of the Environment) for the Province of León.

The amount of herbaceous vegetation remained almost constant during the study period, despite the regional policies applied in the Cantabrian Mountains aimed at increasing pasture areas and enhancing extensive grazing system to economically and ecologically sustainable levels (Celaya et al., 2007). This pattern may result from the compensatory effect of areas where shrub-cutting has been intensive (mainly in the more Atlantic municipalities) and areas where the successional trend linked with the abandonment of agricultural fields is continuing (mainly the most Mediterranean municipalities).

The extent of bare ground also increased over the study period, mainly due to the opening of three large quarries in the area since 1995. Silvicultural measures have also increased the percentage of bare ground in the study area. Shrub clearance has resulted in the affected areas apparently remaining as bare ground before the recovery of pasture species (especially in areas where *Genista* spp. and *Cytisus purgans* have been cleared).

4.2 Land cover change modeling

A general goal of ecologists studying patterns of land cover change is to develop useful predictive models from many possible explanatory variables (e.g. Guisan and Zimmerman, 2000; Suárez-Seoane et al., 2002) to identify drivers of change. These models can be used as tools allowing managers to develop spatially-explicit environmental policies aimed at managing specific types of land cover to maintain biodiversity and the services they provide. In this sense, the binary logistic regression model successfully identified the drivers that have controlled the two main land cover changes under study (forest expansion and loss of shrubs). This knowledge is essential for defining regional strategies in the Cantabrian Mountains. Nowadays, the management of this area is shared among three administrative sections. Land cover changes, ecological processes and their drivers can differ between sections. Different authorities are responsible for each section, meaning that management decisions can be designed and performed independently from the others. However, the new challenges for conservation determined by EU policies will require an integrated understanding of the processes in the protected areas (Natura 2000 sites), whose boundaries do not correspond to administrative sections. Therefore, spatial models like the ones developed in this study can help managers to: i) understand what the main land cover processes in the whole mountain range are and where they are taking place; ii) identify the factors driving those changes as well as the extent at which they are acting; iii) integrate local-focused management and regional plans aimed at preserving areas of natural importance as a whole.

Forest expansion was best explained by topographic variables (altitude and slope), together with variations in the number of goats. Negative standardized logit coefficients indicated that the probability of observing the analyzed trajectory decreases at higher altitudes and slopes. This may be due to the fact that forest recovery has occurred mainly in areas at medium altitude and with low slope close to the valley bottoms, where deeper and more fertile soils are found (Gil and Torre, 2007). The loss of traditional management, based on timber extraction, and the abandonment of agricultural fields, located close to the villages, have favoured an increase in forest in lower areas. However, the logit coefficient for reductions in the number of goats indicated a positive relationship with the probability of forest expansion. Goats use not only herbaceous vegetation, but also woody species (Papachristou et al., 2005), whereas cattle and sheep only use herbaceous vegetation. This fact should be taken into account to understand the role played by goats in keeping both the forest understory clean of ground fuels and grassland areas clean of shrubs and wooded patches (Verdú et al., 2000; Celaya et al., 2007; Sebastián-López et al., 2008).

In contrast, the model for the loss of shrubland cover had low accuracy (AUC=0.62), probably because the change it describes depends not only on biophysical or socio-economic factors, but also on decisions made by managers at a regional level. The opening of pastoral areas by means of shrub clearing has occurred only in municipalities where the neighborhood council demanded them. Shrublands are distributed throughout the study area, across a wide range of environmental conditions. The locations of shrubland clearances are only dependent on the specific place where pastures are required for sheep and cattle grazing and the feasibility of the terrain as a source of short-term pasture. Therefore, even if this land cover change is clearly detected by LANDSAT images, it is difficult to model using logistic regression. The model, in spite of its low accuracy, provides a general description of the main drivers influencing the observed changes. Changes in cattle numbers were the most important variable in the model, reflecting the increased use of pastures for this kind of livestock, which has become more and more popular in the Cantabrian Mountains of León, as in other areas of the Cantabrian range (Rescia et al., 2008). As a result of the abandonment of the transhumance system, summer pastures have progressively been occupied by cows (even from neighboring provinces) due to their easier management: farmers can leave the cattle to roam free in the pastures for days, which imply a lower effort than having sheep or goats. Changes in the number of sheep were also important in the model because, as mentioned above, sheep together with cattle are responsible of the demand for shrub cutting.

5. CONCLUSIONS

After more than fifty years of rural abandonment and loss of traditional management in León Cantabrian Mountains, important land cover changes can be detected: an increase in bare ground, loss of shrublands and increase in forest cover. These trends have become more significant since 2000, when modification in forestry policies introduced important changes aimed at substituting for the loss of traditional management practices (mainly shrub clearance). LANDSAT images were an important tool for analyzing these changes at a regional scale in mountain areas with high heterogeneity. Binary logistic regression models allowed the main drivers linked with these changes to be identified. In general terms, predictive models such as these could be used as a spatially explicit tool to address management strategies. Based on this information, managers can focus their attention on areas with a high risk of change and declare them as priority areas in terms of conservation and management. Knowledge about the drivers of change can also help in developing particular policies aimed at promoting the maintenance of traditional activities, which could be essential for preserving the cultural landscapes in the Cantabrian Mountains.

6. ACKNOWLEDGEMENTS

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***Research Highlights**

- LANDSAT-based classification and modelling identify land cover changes and drivers.
- Forest expansion and loss of shrublands are the major trajectories of change.
- Forest expansion is mostly linked to land abandonment and livestock changes.
- Loss of shrub cover is mostly linked to management decisions (shrub clearance).
- Predictive models can be used as a spatially explicit support tool for land managers.

Figure1
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Figure 1. Location of the study area in the north of León province, Spain.

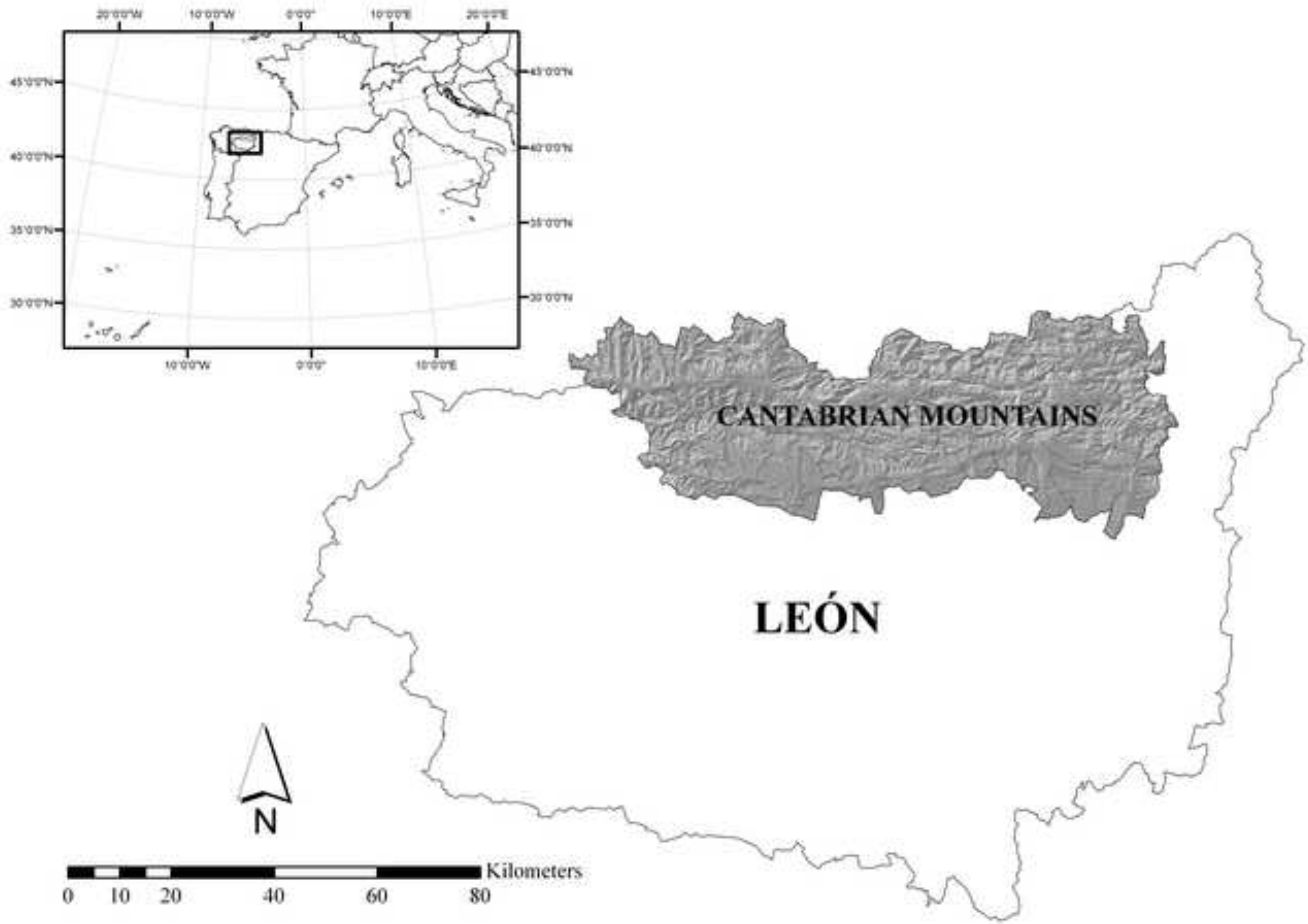

Figure 2
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Figure 2. The four categorical land cover maps derived from the LANDSAT scenes used in the analysis (1991, 1995, 2000 and 2004).

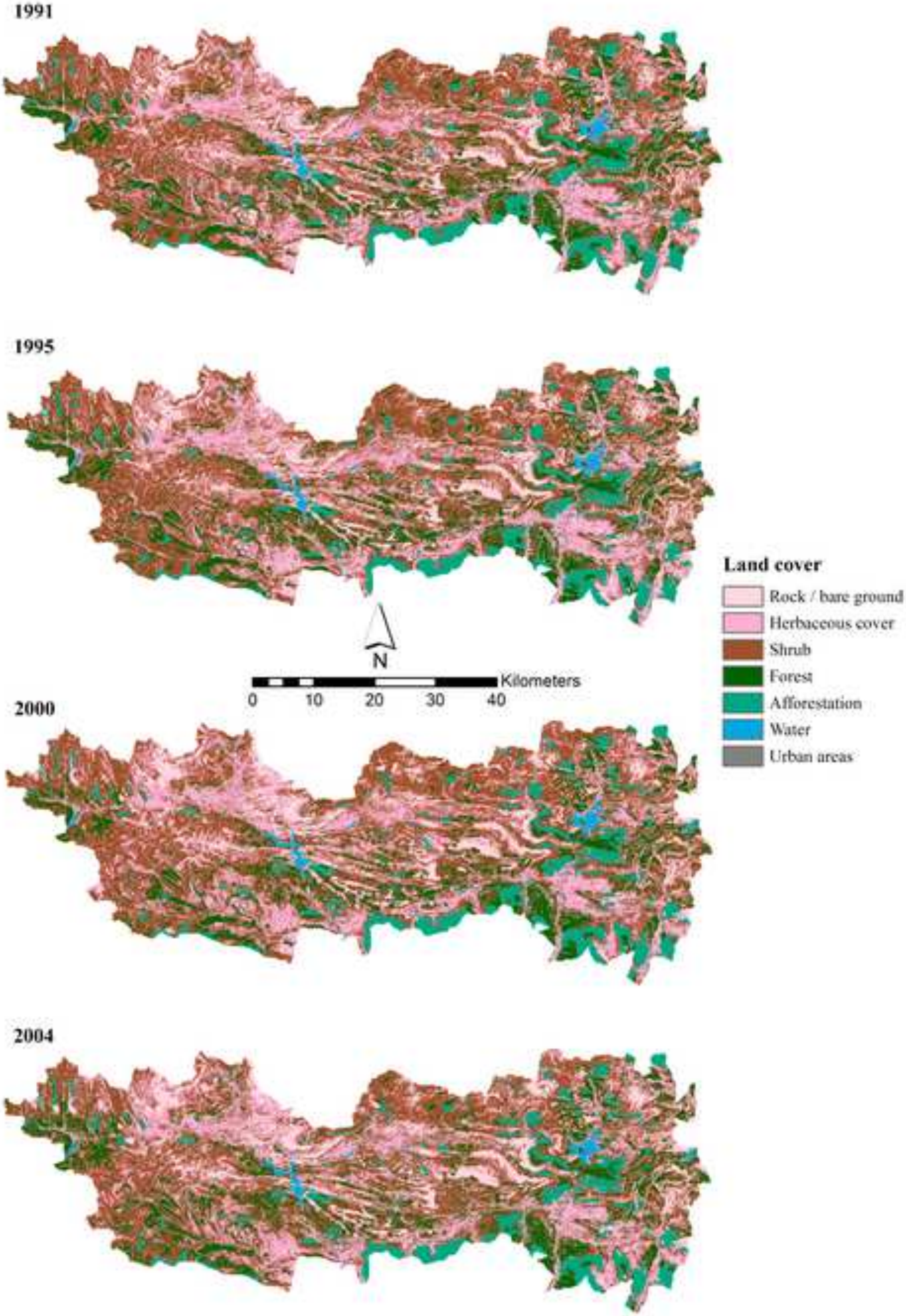

Figure3
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Figure 3. Probability of forest cover expansion

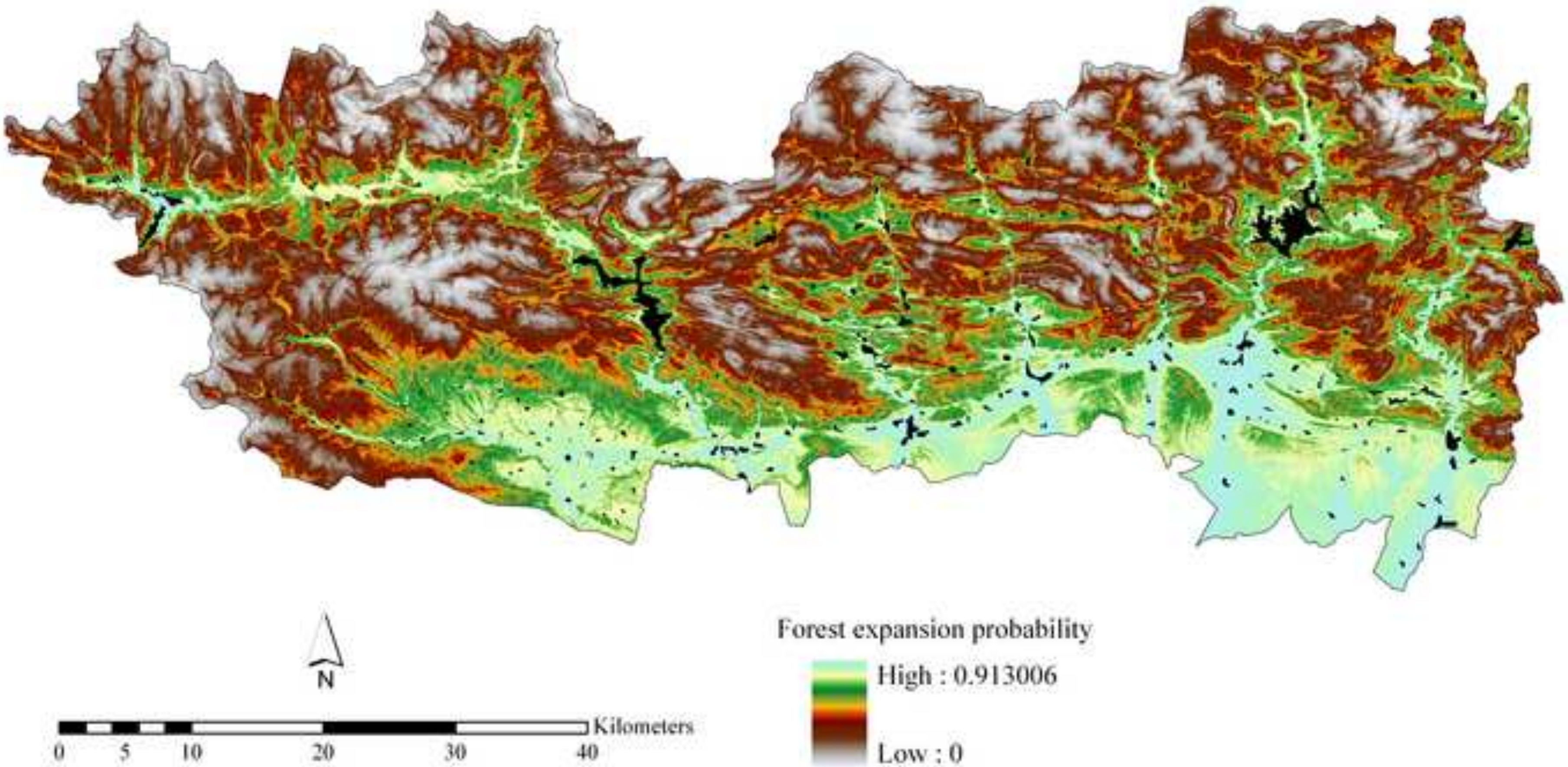

Figure4
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Figure 4. Probability of loss of shrub cover.

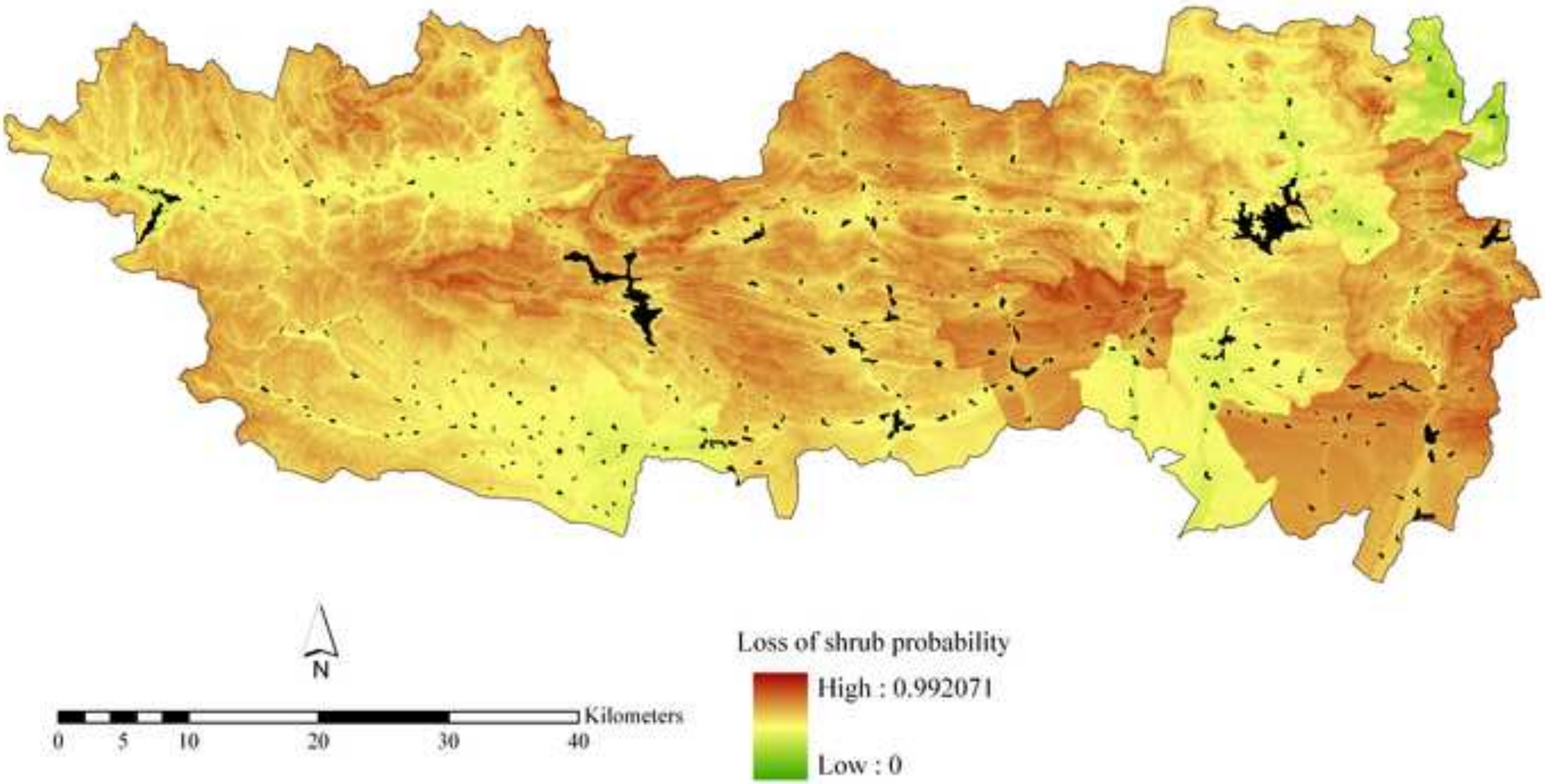

Table 1. LANDSAT images used in the land cover change analysis showing dates of acquirement, type of sensor and sun elevation angle expressed in degrees units (S. E. A).

Date	Sensor	S. E. A
08-04-1991	TM	54.15
08-15-1995	TM	48.87
09-05-2000	ETM+	49.45
09-24-2004	TM	42.72

Table2

Table 2. Data sources and variables used for LANDSAT image classification and analyses of land cover change.

Type	Data source, scale and spatial resolution	Date	Code	Variable definition
Remote sensing variables	LANDSAT images (30m)	1991, 1995 2000, 2004	*Bx*	Bands from 1 to 7 (excluding band 6)
			NDVI	Normalized Differenced Vegetation Index (NDVI), calculated as (NIRC – NR)/ (NIRC + NR) or (BAND 4- BAND3)/ (BAND4 + BAND 3).
			GI	Greenest Index calculates as (NIRC / NG -1) or (BAND 4/ BAND2 – 1)
			TEX	Texture: variable which enhances texture differences between categories. Calculated from the third band of each image.
Topographic variables	Digital Elevation Model (5, 30 m), Junta Castilla y León		*DEM*	Altitude
			SLO	Slope (degrees)
Land cover variables	Vectorial layers at 1:200,000 scale, Junta Castilla y León	2000	*URB*	Urban areas
			WAT	Reservoirs, lakes and pools
		1996	*CON*	Patches of conifer plantation established from 1900 to 1996.
	Spanish Ministry of the Environment (Second and Third National Forest Inventories at 1:200,000 scale)	1996	*NFI2*	Vegetation
		2006	*NFI3*	Vegetation
Socio-economic variables	Spanish Statistics Institute, Caja España	1991-2004	*POP*	Changes in human population density (municipality level) by Ha
	Junta Castilla y León, sanitary campaigns	1999-2005	*CAT* *SHE* *GOA*	Change in number of head of cattle, sheep and goats (municipality level) by Ha

Table3

Table 3. Producer's accuracy (P.A.) and user's accuracy (U.A.) values specified per land cover category and year. The percentages of the total study area occupied each year per land cover category are also detailed. NC91/04 represents the percentage of net change for the period 1991-2004 (note that these percentages of change were calculated on the basis of the surface occupied by each land cover in 1991).

	Rock/bare ground			Herbaceous			Shrub			Forests			
	P.A.	U.A.	%	P.A.	U.A.	%	P.A.	U.A.	%	P.A.	U.A.	%	Overall accuracy
1991	96.12	95.19	11.91	84.16	94.44	21.46	93.33	85.22	42.99	94.34	94.34	13.99	92.05
1995	97.09	96.15	13.37	81.25	89.66	21.03	89.42	86.92	42.41	92.38	88.18	13.55	90.20
2000	87.13	97.78	12.30	78.50	88.42	22.63	96.12	77.95	43.09	90.00	90.91	12.34	87.83
2004	93.88	94.74	14.01	79.57	91.85	21.42	93.50	83.60	40.53	91.00	87.03	14.39	89.10
NC91/04			+17.69			-0.19			-5.72			+2.88	

Table 4. Cross-tabulation matrix analyses between the main land covers (values expressed in percentages).

	2004			
1991	Rock/bare ground	Herbaceous vegetation	Shrubs	Forests
Rock/bare ground	67.36	9.00	8.94	1.55
Herbaceous vegetation	15.90	53.26	16.90	5.94
Shrubs	16.00	27.06	65.42	33.50
Forests	0.74	10.68	8.73	59.01

Table5

Table 5. Main drivers retained in the final BLR models and their estimated coefficients.

	Variable	β_1	SE	Wald	Sig.	Exp (β_1)
Forest expansion	Altitude (*DEM*)	-0.004	0.000	164.352	0.000	0.996
	Slope (*SLO*)	-0.099	0.018	29.491	0.000	0.906
	Goat change (*GOA*)	10.686	5.280	4.096	0.043	43759.825
	Constant	5.787	0.381	230.285	0.000	326.154
Loss of shrub cover	Altitude (*DEM*)	0.001	0.000	9.704	0.002	1.001
	Slope (*SLO*)	0.044	0.012	13.104	0.000	1.045
	Cattle change (*CAT*)	5.332	1.689	9.965	0.002	206.751
	Sheep change (*SHE*)	-3.781	1.063	12.657	0.000	0.023
	Constant	-1.313	0.311	17.824	0.000	0.269

β1: Coefficient, SE: Standard error of estimate, Exp (β1): Exponential coefficient/odds ratio