


Woody plant encroachment following grazing abandonment in the subalpine belt: a case study in northern Spain

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Abstract The montane and subalpine belts in European mountains were affected by intense land use/land cover changes during the twentieth century. In the case of the subalpine belt, most European mountains were affected by complex deforestation processes from Neolithic times, leading to the expansion of summer grasslands to support grazing by transhumant sheep and goats flocks. This resulted in an altitudinal reduction of the tree line and the occurrence of severe soil erosion and shallow landsliding. The intense livestock pressure over centuries explains why the landscape remained without major change until the middle of the twentieth century. Since then, depopulation, land abandonment, and the disappearance of the transhumant system have resulted in a marked decline in livestock numbers and subsequent encroachment of shrubs and trees in the subalpine belt. The

Urbión Mountains (Iberian Range, northern Spain) provides one of the clearest examples in the Iberian Peninsula, where there was intense deforestation since the Neolithic period to enlarge the area of subalpine summer grasslands. The recent reversal of this situation in the last decades led to (i) a marked trend to dense forest, which resulted in an average 200 m altitudinal advance, and (ii) the spatial contraction of shrublands, which have been replaced by dense forest in the highest areas. Changes will continue in the future, possibly enhanced by global warming.

Keywords Woody encroachment · Subalpine belt · Deforestation · Land use/land cover changes · Land abandonment

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Introduction

European mountains have been subject to intense land use changes during the last two centuries (MacDonald et al. 2000; Tasser and Tappeiner 2002; Poyatos et al. 2003; Vicente-Serrano et al. 2004; Améztegui et al. 2010; García-Ruiz et al. 2010; Arnáez et al. 2011; García-Ruiz and Lana-Renault 2011; Gouveia et al. 2016) because of population changes and societal and economic interactions with the lowlands. In most Mediterranean mountains, the highest population density was recorded in the middle of the nineteenth century, when hillslopes and valley bottoms supported an integrated land use system aimed at ensuring the survival of local populations through cultivation of all possible areas. Transhumance of large sheep flocks forced the transformation of the upper montane and subalpine belts into grasslands, to enable the enlargement of summer grazing areas (García-Ruiz and Lasanta 1990). Consequently, in the middle of the nineteenth century, most European mountains were under extreme human pressure including (i) deforestation of most slopes above 1500 m a.s.l. to enlarge the area covered by subalpine grasslands, (ii) frequent use of fire to control shrub and forest expansion, (iii) the presence of large (in some cases transhumant) sheep flocks, and (iv) the occurrence of a variety of erosion processes (sheet wash, rilling and gulying, landslides) (Höllermann 1985; del Barrio and Puigdefábregas 1987; Tasser et al. 2003; García-Ruiz et al. 2010, 2015; Lana-Renault et al. 2011; García-Ruiz 2015); these erosion processes were responsible for fluvial channel aggradation (Gómez-Villar et al. 2014) and alluvial fan enlargement (Gómez-Villar et al. 2006). In the Mediterranean and central European mountains, the subalpine belt had been deforested since Prehistoric times, as confirmed by the presence within the soil or in peat bogs of charcoal from the Neolithic period and the Bronze and Iron ages (e.g., Colombaroli et al. 2008; Guiguet-Covex et al. 2011; Cunill et al. 2012; Lasheras-Álvarez et al. 2013; Pérez-Sanz et al. 2013). This was also the case for the Iberian Range (northern Spain), where the subalpine belt of the Urbión Mountains has been deforested since the Neolithic (García-Ruiz et al. 2016). This process caused a marked lowering of the altitude of the tree line below its natural elevation, which is estimated from other mountains of the Iberian Peninsula to have been an average of 400 m (e.g., Höllermann 1985; García-Ruiz et al. 1990).

Depopulation and land abandonment have been the most significant changes affecting Mediterranean mountains since the mid-twentieth century, although it was far from uniform. The peak of depopulation in the central and west European mountains occurred at the beginning of the twentieth century (Estienne 1989; Diry 1995). However, in many Mediterranean mountains, particularly on the Iberian Peninsula, depopulation was a slow process from the end of the nineteenth century, followed by a rapid acceleration that peaked during the 1960s

and 1970s, in parallel with a dramatic process of farmland abandonment (Lasanta 1990; Vitte 1992). Since that time, most of the cultivated hillslopes have been abandoned, with the only remaining cultivated areas being the valley bottoms, large alluvial fans, and some perched flats where cereal crops have been replaced by mown meadows (García-Ruiz and Lasanta 1990). The almost total disappearance of transhumant flocks has led to the abandonment of many summer subalpine pastures. Many aspects of mountain Mediterranean landscapes are changing, including runoff generation (Beguería et al. 2003; Viviroli et al. 2003; García-Ruiz et al. 2011), soil erosion and landsliding (Bathurst et al. 2007; García-Ruiz et al. 2010), fluvial dynamics (Liébault et al. 2005), the spatial organization of plant cover, and the ecosystem services that the area provides to local communities and nationally (Egoh et al. 2007; Egarter Vigl et al. 2016). Major changes have been detected in the Alps (e.g., Tasser and Tappeiner 2002) and in the Pyrenees (e.g., Lasanta-Martínez et al. 2005; Lasanta and Vicente-Serrano 2007; Gartzia et al. 2014). Nevertheless, changes in land cover of the subalpine belt have not been so intensively studied, although they could be extremely important from the point of view of snow accumulation and water resources (López-Moreno 2005).

The Urbión Mountains have a long history of landscape transformation. The development of sheep transhumance was extraordinary and led to enlargement of the subalpine grasslands, generalized deforestation, and the occurrence of intense landsliding at altitudes above 1500 m a.s.l. (García-Ruiz et al. 2016). The recent and rapid decline of the transhumant system, with the consequent decline in sheep flocks, has resulted in rapid encroachment by shrubs and forest. In this study, we investigated this process in the Urbión Mountains, and more precisely in the Ormazal Valley. The study areas is an example of what is happening in the subalpine belt of mountains of northern Spain, with the aim of identifying those factors explaining the spatial complexity of woody encroachment during the period 1956–2006.

Study area

The Urbión Mountains is one of the highest massifs (peaking at 2228 m) of the Iberian Range, in the northern Iberian Peninsula (30N 514118 4657163). The massif is mainly composed of Upper Jurassic and Lower Cretaceous small quartz conglomerate, red mudstone, and limestone arranged in a monoclonal relief.

The study was carried out in the Ormazal Valley, which has an area of 2513.8 ha and an altitudinal range of 1240–1887 m a.s.l. The landscape is heterogeneous, ranging from the lower montane belt (cultivated and abandoned fields) to the subalpine belt (grasslands and forests). The valley has relatively gentle slopes (in general, < 30%) and wide divides.

Weather stations are located on the southern face of the massif, at approximately 1100–1150 m a.s.l. A temperature increase has been recorded in the last 60 years; thus, the mean annual temperature at Vinuesa was 7.7°C for the period 1957–1965 and 9.6°C for the period 1990–2001, with an average increase in winter of 1.3°C (García de Celis et al. 2008). Mean annual precipitation exceeds 900 mm above 1000 m a.s.l. and is probably 1500–1600 mm in the main divide, based on an estimate for the neighboring Demanda Sierra (Arnáez Vadillo 1987). Precipitation has declined slightly since the mid-twentieth century, particularly between November and March (García de Celis et al. 2008).

The plant cover is dominated by extensive *Pinus sylvestris* forests on the south-facing slopes, whereas the northern faces are complex landscapes involving *P. sylvestris* and *Fagus sylvatica* forests, grasslands, and shrubs including *Calluna vulgaris*, *Erica cinerea*, *Vaccinium myrtillus*, and *Juniperus communis*. The area has been used in summer for transhumant sheep herds, with two periods of particularly intense activity: the fifteenth century, and the period between the end of the seventeenth and the end of the eighteenth centuries. In 1792, two villages located close to the Ormazal Valley (Viniestra de Abajo and Viniestra de Arriba) accounted for 17,169 (262 per km²) and 19,100 (497 per km²) sheep, respectively (Moreno Fernández 1996). The number of sheep has declined and now accounts for 840 and 2690, respectively. Population has also declined in both villages: Viniestra de Abajo had 689 inhabitants in 1842 and 97 in 2011; Viniestra de Arriba had 418 inhabitants in 1842 and 49 in 2011. Below 1500 m, a large part of the hillslopes was cultivated with cereals, although most of fields were abandoned by the middle of the twentieth century, except for some fields located in the valley bottoms close to the villages.

Methods

Aerial photographs of the Ormazal Valley from 1956 (black and white) and ortho-photographs from 2006 (color), all with a scale of 1:18,000, were used to analyze the evolution of land use and land cover (LULC). In order to reduce methodological inconsistencies in the visual interpretation of two qualitatively different sets of images, a first LULC map was produced based on the best set (2006). A second LULC map was then produced for 1956 using the first one as a base layer, so the interpretation was focused in the changes occurred with respect to this base map. Two maps (one for 1956 and one for 2006) were thus generated, including the following LULC classes: (i) cultivated fields, (ii) abandoned fields, (iii) grasslands, (iv) shrublands, (v) clear forest (where voids up to 20 m occur between trees), (vi) dense forest (where the canopy is continuous, without voids between trees), (vii) eroded areas, (viii) rock outcrops and debris accumulations, and (ix)

reforestation areas. Cultivated and abandoned fields were clearly discriminated visually due to the existence of linear features delimiting the fields. Grasslands, shrublands, and forests were also clearly identified, as were rock outcrops and debris accumulations. The distinction between clear and dense forests was somehow more subjective, so reference areas were first set and used therefore as discriminating patterns. Delimitation of eroded areas within the grasslands was the most difficult due to their small size and diffuse spatial limits, although reference areas were also set and used for discrimination.

The 1956 and 2006 LULC maps were overlapped to assess changes that occurred between the 2 years. Measurement of the areas where changes or no changes had occurred enabled computation of a LULC transition matrix.

To determine the influence of environmental factors on the changes observed in LULC categories, a set of topographic variables was derived from a digital elevation model (Regional Government; <http://iderioja.larioja.org/>) having a spatial resolution of 18 × 18 m and using ArcGIS Desktop. The variables included the following: (i) altitude, (ii) slope gradient, (iii) slope aspect, (iv) distance to the hydrographic network, (v) slope shape (curvature), (vi) solar irradiation, and (vii) the topographic index (Beven and Kirkby 1979).

Discriminant analysis was used to interpret the influence of the variables on the main LULC transitions observed. This involved conversion of the LULC maps to raster format at the same resolution as the topographic data. Discriminant analysis provided information on the topographic variables (independent variables), which enabled differentiation of the groups of pixels classified from the LULC transitions (dependent variables). In total, 16 changes representing 96.3% of all changes recorded were selected for analysis.

Multinomial logistic regression analysis was used to explain the transition of grasslands and shrublands (the two LULC classes most abundant in 1956, and those that experienced the highest rates of change) to other LULC categories. Multinomial logistic regression facilitated calculation of the probabilities of permanence (no LULC change) and change, as a function of covariates (independent variables). The topographic variables selected as covariates were elevation, slope gradient, solar irradiation, and topographic index. The significance of each covariate as an explanatory factor was assessed using the *t* test at a significance level of $\alpha = 0.05$, and the significance of the overall model was assessed using the likelihood ratio test. The calculations were performed using the package *mlogit* version 0.2-4 for the R environment (Croissant 2013). The results were expressed as expected transition probabilities corresponding to average topographic conditions and as changes in these probabilities as a function of unit changes in the topographic variables. Maps were computed from the multinomial models to visually assess the spatial variability of transition probabilities.

Results

Spatial distribution and changes in land use and land cover between 1956 and 2006

The maps in Fig. 1 show the spatial distribution of the LULC categories in 1956 and 2006, and the areas covered by each LULC category in the 2 years is shown in Table 1. By 1956, some cultivated fields (0.4% of the total study area) remained in the lowest part of the catchment (1300–1500 m a.s.l.), adjacent to fields that had been cultivated some decades previously and then abandoned (58.2 ha; 2.3%). The main LULCs corresponded to grasslands (910.6 ha; 36.2%) and shrublands (853.2 ha; 33.9%). Dense forests occupied only 43.6 ha (1.7%), mainly in the northernmost part of the catchment (small patches of *F. sylvatica* forest), and clear forests (*Quercus pyrenaica* and *P. sylvestris*) occupied 231.1 ha (9.2%), scattered throughout shrublands areas. Eroded areas were generally located over bare limestone thinly covered by very open grasslands and occupied 405.3 ha (16.1%).

Substantial changes had occurred by 2006:

1. The expansion of dense forests at the expense of the other LULC categories (particularly grasslands and shrublands) was the main change that occurred in the study area between 1956 and 2006, while the eroded area remained relatively constant.
2. Cultivated fields had been largely abandoned.

Table 1 Land use/land cover (LULC) classes in 1956 and 2006

LULC	1956 (ha)	1956 (%)	2006 (ha)	2006 (%)
Cultivated fields (CF)	10.07	0.4	0	0
Abandoned fields (AF)	58.23	2.3	59.99	2.4
Reforestation (RF)	0	0.0	70.46	2.8
Rock outcrops, debris (R)	1.72	0.1	1.04	< 0.1
Eroded areas (EA)	405.33	16.1	438.37	17.4
Grasslands (pastures) (G)	910.55	36.2	681.97	27.1
Shrublands (S)	853.15	33.9	651.39	25.9
Clear forest (CF)	231.12	9.2	174.18	6.9
Dense forest (DF)	43.64	1.7	436.41	17.4
Total	2513.81	100	2513.81	100

3. Grasslands were still the most extensive land cover (682.0 ha; 27.1% of the total), although there had been marked spatial shrinkage.
4. Shrublands were the second largest land cover (651.4 ha; 25.9%), although they also had declined in area.
5. The clear forest area had decreased slightly, occupying 174.2 ha (6.9%) in 2006, and the area of dense forests had remarkably increased to 436.4 ha (17.4%).
6. A small enlargement of the dense *F. sylvatica* forests was also detected in the north of the catchment.
7. The eroded areas remained relatively constant in 2006, with 438.4 ha (17.4%).

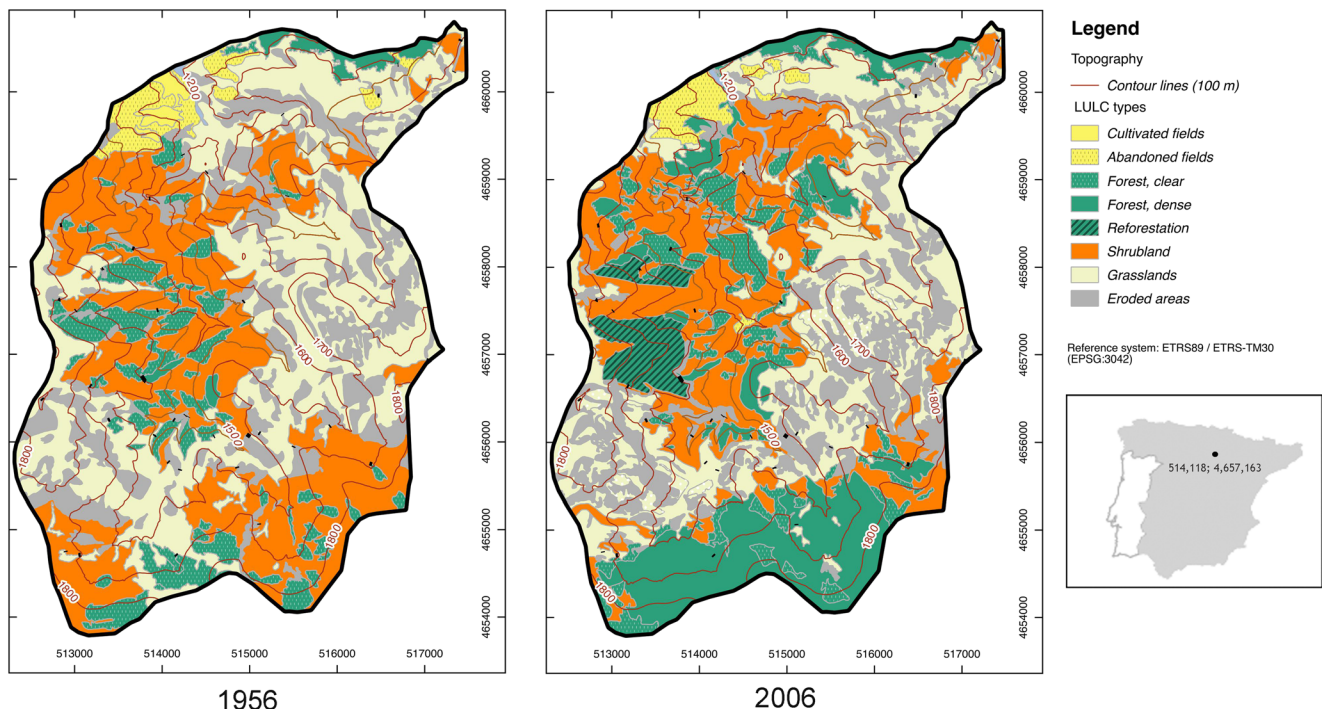


Fig. 1 Maps with LULC categories in the Ormazal Valley in 1956 and 2006 and location of the Ormazal valley in Spain

8. Major changes had occurred in the southernmost part of the study area, where shrublands and clear forests were substituted by a dense forest of *P. sylvestris*. A close spatial relationship was found between the locations of summer grasslands and the eroded areas.

The transition matrix (Table 2) shows the changes that occurred in the area occupied by each LULC category. The most outstanding results shown in this table are described as follows:

1. The average rate of change (approximately 50%) was very high, confirming that the area was affected by highly dynamic changes in LULC categories.
2. Dense forest can be considered a stable or absorbing state of the system, as shown by the high rate of permanence of this LULC category (94.5%).
3. For grasslands, 52.5% maintained their status, with the main transitions being to eroded areas (20.1%) and shrublands (17.3%).
4. For eroded areas, 54.8% remained in the same category and their most common transition was to grasslands. This indicates that a reversible transition occurred between grasslands and eroded areas.
5. Shrubbylands represent a clear transition state. Their permanence rate was relatively low (43.2%), and their transition tended to be to forest (39.4%, including the two forest types).
6. Abandoned fields showed a relatively high rate of permanence (60.4%), with notable transition to grasslands (18.8%) and dense forests (12%).
7. The lowest rate of permanence corresponded to clear forest (12.3%), which transitioned to dense forest (40.9%) and shrublands (32.4%).

Table 2 Transition matrix between LULC classes in 1956 (vertical) and 2006 (horizontal). Values are expressed in percentage over the area covered by each LULC class in 1956. CF, cultivated fields; AF, abandoned fields; RF, reforestation; R, rock outcrops and debris; EA, eroded areas; G, grasslands; S, shrublands; CF, clear forest; DF, dense forest. The diagonal numbers in italics indicate the percentage of each LULC category that did not change

1956	2006									
	CF	AF	RF	R	EA	G	S	CF	DF	Total
CF	<i>0</i>	99.6	0	0	0	0	0	0	0.4	100
AF	0	<i>60.4</i>	0	0	4.4	18.8	4.3	0	12	100
RF	0	0	<i>0</i>	0	0	0	0	0	0	0
R	0	0	0	<i>0</i>	0	0	0	0	100	100
EA	0	0	0.4	0.1	<i>54.8</i>	26.9	11.6	4.7	1.5	100
G	0	1.3	0.4	0	20.1	<i>52.5</i>	17.3	2.5	5.9	100
S	0	0.3	4.3	0.1	3.2	9.6	<i>43.2</i>	12.2	27.2	100
CF	0	0.1	12.6	0	1.3	0.4	32.4	<i>12.3</i>	40.9	100
DF	0	0	0.2	0	1.1	2.7	1.1	0.5	<i>94.5</i>	100

8. In 2006, a new category of LULC (reforestation) was identified; this category mainly incorporated former clear forests (12.6%) and shrublands (4.3%).

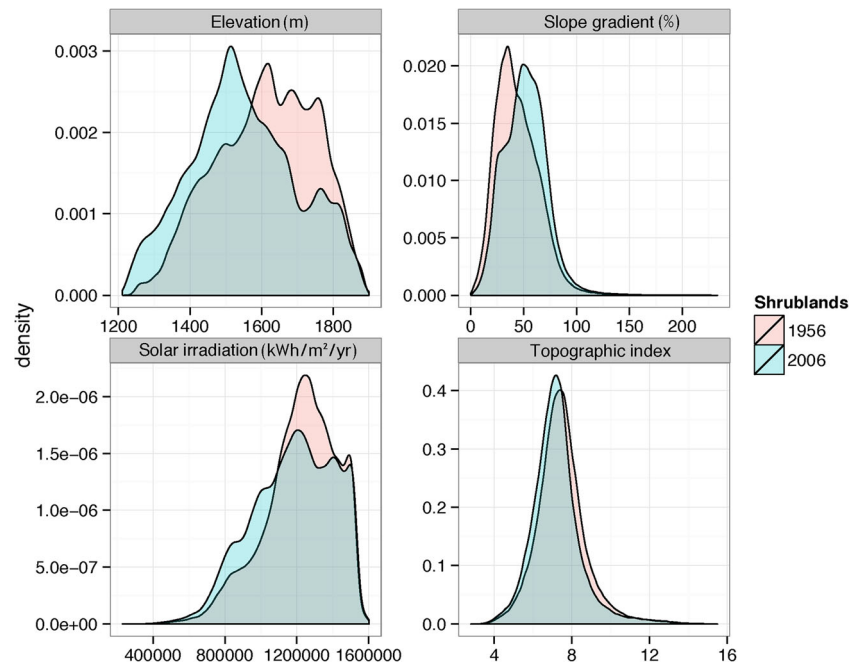
Factors controlling land use and land cover transitions

Figures 2 and 3 show the frequency distribution of four topographic variables (altitude, slope gradient, solar irradiation, and topographic index) for shrublands and forest in 1956 and 2006. It is noteworthy that neither summer grasslands nor eroded areas significantly changed their “topographic signature” between the 2 years, other than a small increase in altitude in the case of summer grasslands. Conversely, shrublands (Fig. 2) and dense forests (Fig. 3) showed major changes. Thus, while shrublands tended to be relegated to areas of lower altitude, higher slope gradient, and lower levels of irradiation, the dense forests colonized higher altitudes, shallower gradients, and areas affected by higher levels of solar irradiation. This suggests that the substitution of shrublands by dense forests was markedly influenced by topographic factors. The average altitude for dense forest was 1448 m in 1956 and 1638 m in 2006 (an altitudinal advance of almost 200 m). In the case of shrublands, the average altitude was 1627 m in 1956 and 1575 m in 2006.

To assess the influence of topographic variables on the main LULC changes we performed a discriminant analysis. The change from shrublands to dense forest was the largest in area (15.4%), followed by the changes from summer grasslands to eroded areas (12.7%), from summer grasslands to shrublands (10.3%), from shrublands to clear forest (8.3%), from eroded areas to summer grasslands (8.3%), and from clear forest to dense forest (7.2%). The discriminant analysis identified three functions that differentiated the selected LULC transitions. An analysis of the contrast of the functions through the Wilks’ lambda and the chi-square analysis showed a significance level < 0.05 , indicating that the model could distinguish among the LULC transition groups.

Figure 4 shows the results of the discriminant analysis and enables interpretation of the topographical factors explaining most of the major LULC changes in the study area. Function 1 (horizontal axis) strongly correlated with the altitude, clearly discriminated changes in the agricultural areas (cultivated fields to abandoned fields, and abandoned fields to summer grasslands), with negative values for the centroids, and discriminated changes to shrublands, clear forest, and summer grasslands, which became dense forest (positive values). Function 2 (vertical axis) was defined by slope aspect and to a lesser extent by the slope gradient. This function discriminated changes from clear forest to eroded areas (positive values), opposed to changes recorded in grasslands (summer grasslands to clear forest and dense forest). Based on these

Fig. 2 Density curves for 1956 and 2006 showing the frequency distribution of shrublands as a function of elevation (m), slope gradient (%), solar irradiation ($\text{kWh m}^{-2} \text{ year}^{-1}$), and topographic index (unitless)



results, LULC changes in the Urbión Mountains between 1956 and 2006 were influenced by topographic variables. The discriminant analysis indicated that the previously cultivated areas, located at the lowest altitudes (1300–1400 m), were in 2006 occupied by abandoned fields and summer grasslands. Conversely, at altitudes of 1600–1700 m the shrublands had transitioned to eroded areas, or more commonly to dense forest. Areas of summer grassland and clear forest in 1956 transitioned to dense forest in 2006. Solar irradiation (which depends on slope aspect) and hillslope gradient also

discriminated transitions in LULC, particularly the colonization of summer grasslands by dense forest (preferentially on hillslopes having gradients of 14–15° and southwestern exposures), and the degradation of clear forest to eroded areas (on gradients of almost 20° and eastern exposure).

Application of multinomial logistic regression analysis to the areas covered by grasslands in 1956 enabled estimation of the permanence and probability of transition, as a function of topographic covariates. The model was significant overall (likelihood ratio test p value < 0.05), indicating a degree of

Fig. 3 Density curves for 1956 and 2006 showing the frequency distribution of forests as a function of elevation (m), slope gradient (%), solar irradiation ($\text{kWh m}^{-2} \text{ year}^{-1}$), and topographic index (unitless)

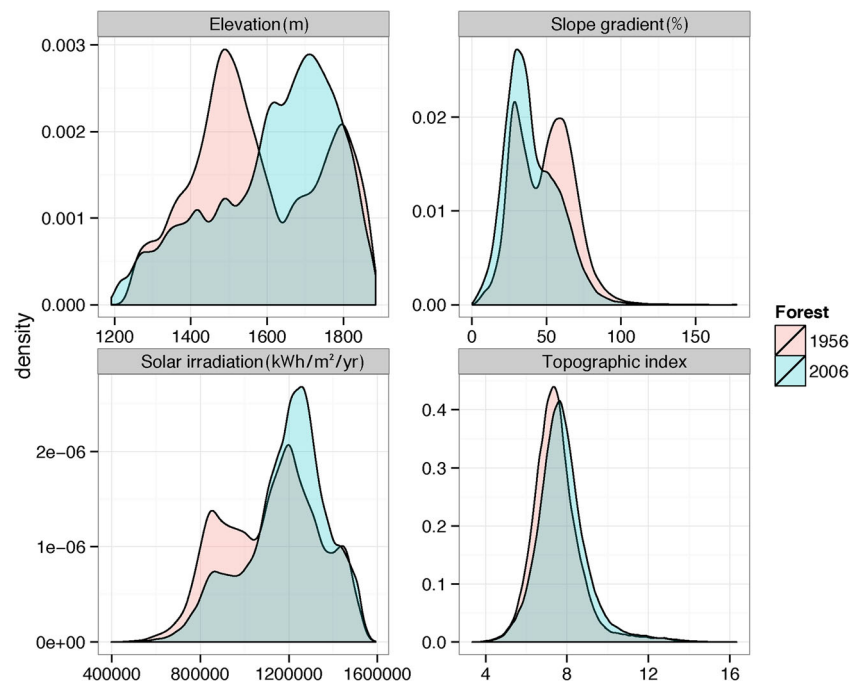
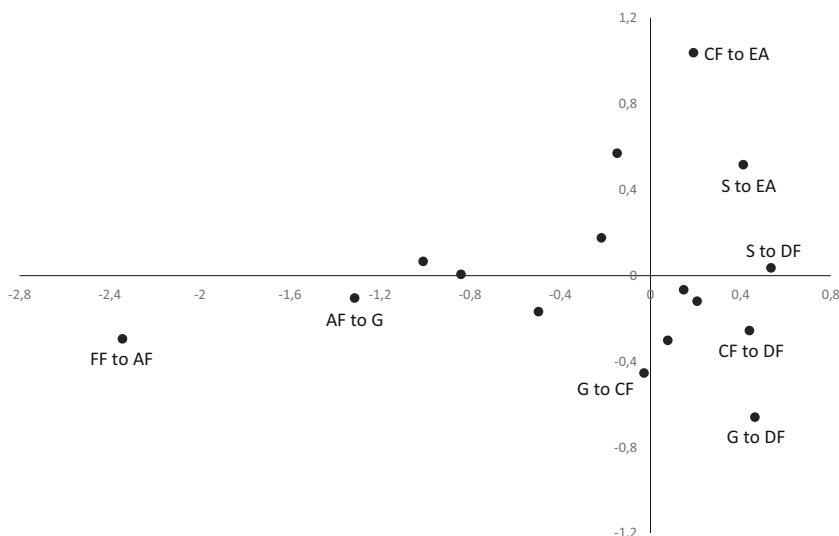


Fig. 4 Centroids in the discriminant analysis. Horizontal axis (function 1); vertical axis (function 2). CF to AF, cultivated fields to abandoned fields; AF to G, abandoned fields to summer grasslands; CF to EA, clear forest to eroded areas; S to EA, shrublands to eroded areas; S to DF, shrublands to dense forest; CF to DF, clear forest to dense forest; G to CF, summer grasslands to clear forest; G to DF, summer grasslands to dense forest



topographic control, although only the slope gradient and solar irradiation were found to be significant according to the *t* test; the other topographic variables (elevation, topographic index) were not significant and thus removed from the model.

The transition probabilities were consistent when compared with the observed transition matrix (Table 3), although the probability of permanence was slightly overestimated (62 vs. 53%), and the probability of transition to eroded areas was underestimated (16 vs. 20%). A slight overestimation of the permanence rate was expected, as it was the reference level with which the other transitions were compared.

The influence of topographic variables on the transitions from grassland is summarized in Table 4. With respect to the transition to forest, each increase of 10° in slope gradient resulted in an increase of 3.7% in the probability of transition, while an increase of 200 kWh m⁻² year⁻¹ in solar irradiation reduced the probability by 5.6%. For the transition to eroded areas and shrublands, increasing slope gradient and solar irradiation had stronger positive effects than occurred for forests.

Application of the resulting model to the spatially explicit topographic variables enabled computation of transition probability maps (Fig. 5). For each grid cell that was occupied by grasslands in 1956, these show the probability of grassland

permanence and transition to another LULC category by 2006, as estimated using the regression model. The spatial distribution of transition probabilities varied in accordance with variation in the topographic characteristics.

Application of the same analysis to the areas covered by shrublands in 1956 showed an even stronger correlation of permanence and transition probabilities with the observed transition rates (Table 3). Overprediction of the probability of permanence was reduced (48% vs. the observed rate of 43%), and only the probability of transition to forest was slightly overpredicted (42 vs. 39%). The topographic conditions were only significant for the transition to forest but revealed a strong influence. For instance, every change of 10° in the slope gradient resulted in an 11.4% increase in the probability of transition to forest, while an increase of 200 kWh m⁻² year⁻¹ in irradiation decreased the probability by 7.7% (Table 4). The transition probability maps (Fig. 6) reflect this strong topographic control. For instance, the probability of shrubland permanence was highest for the mid-altitude slopes in the northwest of the study area, where deforestation and land abandonment had led to the occurrence of shrubs and clear forest. In contrast, the probability of transition from shrublands to forest was highest for the steep and shady slopes in the southernmost part of the study area and

Table 3 Permanence and transition probabilities of grassland and shrubland areas in 1956, observed and predicted by multinomial logistic regression analysis

	Probability of permanence	Probability of transition to forest	Probability of transition to eroded	Probability of transition to shrublands
Grasslands				
Predicted	0.62	0.07	0.16	0.15
Observed	0.53	0.08	0.20	0.17
Shrublands				to grasslands
Predicted	0.48	0.42	0.04	0.06
Observed	0.43	0.39	0.03	0.10

Table 4 Transition probabilities from grasslands and shrublands to (i) forest, (ii) eroded, and (iii) shrubland or grassland, considering average topographic conditions, and variation in the transition probabilities as a function of changes in the topographic variables (only significant effects)

From grasslands...	... to forest	... to eroded	... to shrublands
Average topographic conditions (36%, 1229 kWh m ⁻² year ⁻¹)	10.4%	20.3%	20.0%
Slope gradient + 10%	+ 3.7%	+ 15.8%	+ 13.8%
Solar irradiation + 200 kWh m ⁻² year ⁻¹	- 5.6%	+ 23.7%	+ 6.7%
From shrublands...	... to forest	... to eroded	... to grasslands
Average topographic conditions (36%, 1223 kWh m ⁻² year ⁻¹)	46.5%	7.4%	11.3%
Slope gradient + 10%	+ 11.4%		
Solar irradiation + 200 kWh m ⁻² year ⁻¹	- 7.7%		

also on the low altitude and shady slopes close to formerly cultivated areas in the northwest of the study area. In both areas, there was also a relatively high probability of transition to grasslands. The transition to eroded areas, besides being very low overall (less than 5%), indicates the instability of shrublands in the northwestern sector of the study area, where the steepest slopes are located, and close to areas affected by human-induced reforestation.

Discussion

Major changes have occurred in the montane and subalpine belts of the European mountains. In the Urbión Mountains, northern Iberian Peninsula, forest clearing began in Neolithic times and has continued to the present (García-Ruiz et al. 2016). Forest clearing favored the expansion of transhumant sheep flocks and caused (i) altitudinal lowering of the tree line by at least 400 m and (ii) the triggering of soil erosion processes and shallow landsliding. The latter is based on the presence of large landslide scars and areas with bare soil, which in other mountain areas including the Pyrenees (García-Ruiz et al. 2010) and the Alps (Tasser et al. 2003; Guiguet-Covex et al. 2011) have been attributed unequivocally to deforestation. In the Pyrenees, a palinological study of Tramacastilla Lake (Gállego Valley) confirmed the occurrence of moderate erosion during small deforestation events approximately 4000 cal years BP (Montserrat-Martí 1992) and intense soil erosion in the subalpine belt since the Middle Ages, following extensive fires that coincided with the maximum period of transhumance.

Dating of charcoal fragments in the soil in the Urbión Mountains has suggested that forest clearing and grazing in the subalpine belt has been a common practice since the Neolithic and Calcolithic periods, and also during the Bronze, Iron, and Middle ages (García-Ruiz et al. 2016). Subsequently, the deforested landscape and the large areas of summer grasslands were maintained because of the presence of thousands of sheep grazing from the beginning of June to the end of September. By the middle of the twentieth

century, aerial photographs show that most of the study area had been deforested, and only small patches of dense forest remained in some valley bottoms, indicating that the traditional landscape was the consequence of intense human occupation and management. Apart from a small area dedicated to farmland activity, the remainder of the study area showed the effects of grazing, with clear forests, shrublands, summer grasslands, and eroded areas. These latter were located in close spatial proximity (and similar topographic signatures) to summer grasslands, suggesting that soil erosion was a direct consequence of deforestation.

In 2006, 50 years later, the landscape retained some of the original characteristics, although major changes had occurred. The main reason for the changes was the decline in livestock numbers that had occurred in the preceding few decades, and the subsequent decrease in livestock pressure. This led to a relatively complex process of woody encroachment characterized by (i) the expansion and average 200 m altitudinal advance of dense forests at the expense of other LULC classes, which is consistent with tree lines being highly sensitivity to environmental changes (Holtmeier and Broll 2005; Wallentin et al. 2008; Pecher et al. 2011; Tasser et al. 2017); (ii) the spatial contraction of shrublands, which were substituted by dense forests at the highest altitudes; (iii) shrublands and clear forest transitioning towards dense forest; and (iv) a change in LULC categories affecting approximately 50% of the study area during the study period, confirming that the landscape configuration was a consequence of human pressure, through livestock grazing, and domestic wood consumption.

Interestingly, it was found that a reversible transition occurred between erosional features (mostly due to shallow landsliding) and pastures, implying that the eroded stage was not a final stable state, at least within the time frame of this study. This result, however, has to be taken with care since the distinction between pastures and eroded pastures in the aerial photographs was not devoid of difficulties, especially in the oldest set (1956).

The increase in altitude of the tree line may also be attributed, at least in part, to the recent increase in temperature, as reported by García de Celis et al. (2008) for the southern

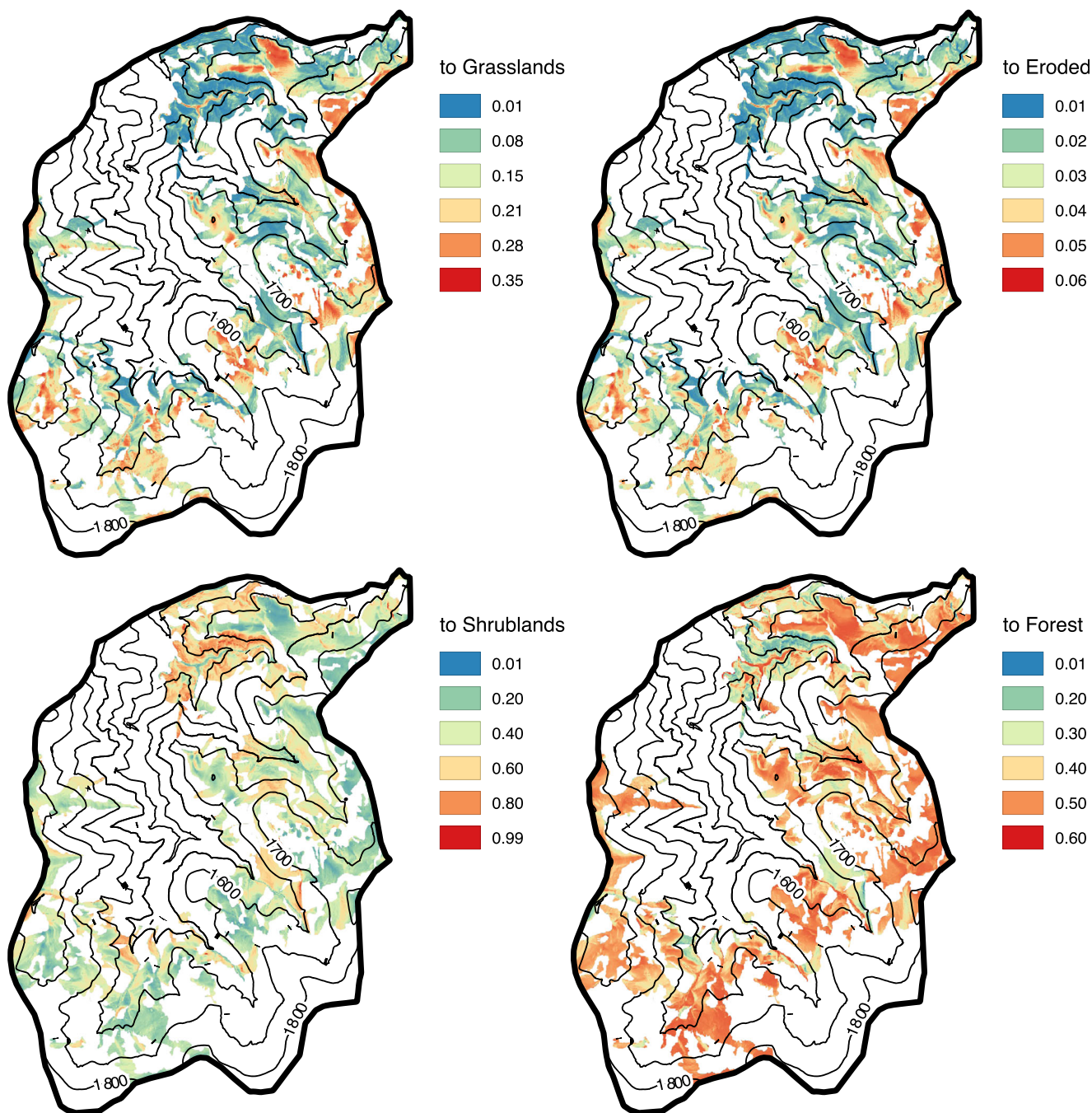


Fig. 5 Transition probabilities for the areas covered by grassland in 1956: grassland permanence and transition to forest, eroded areas, and shrubland

slopes of the Urbión Mountains. Nevertheless, the irregular distribution of forest and summer grasslands in Ormazal Valley suggests that forest expansion has been largely a consequence of grassland abandonment. The effect of global warming on forest development can only clearly be detected in the natural upper forest limit, following the altitudinal reduction in the tree line that occurred during the Little Ice Age (Camarero et al. 2015).

Woody encroachment in the subalpine belt has also been reported in many other European mountains. For instance,

densification of forest near the tree line has been reported in the eastern Pyrenees (Batllori and Gutiérrez 2008), where the tree line has moved upwards by an average of almost 40 m because of land use changes (Améztegui et al. 2015). In the central Pyrenees, Gartzia et al. (2014) reported the expansion of shrublands and forest at the expense of summer grasslands. These authors noted the important role of the distance to patches of woody species in explaining the pattern of tree recruitment. In the Ijezu Valley, in the central Pyrenees, Sanjuán et al. (2016) noted the expansion

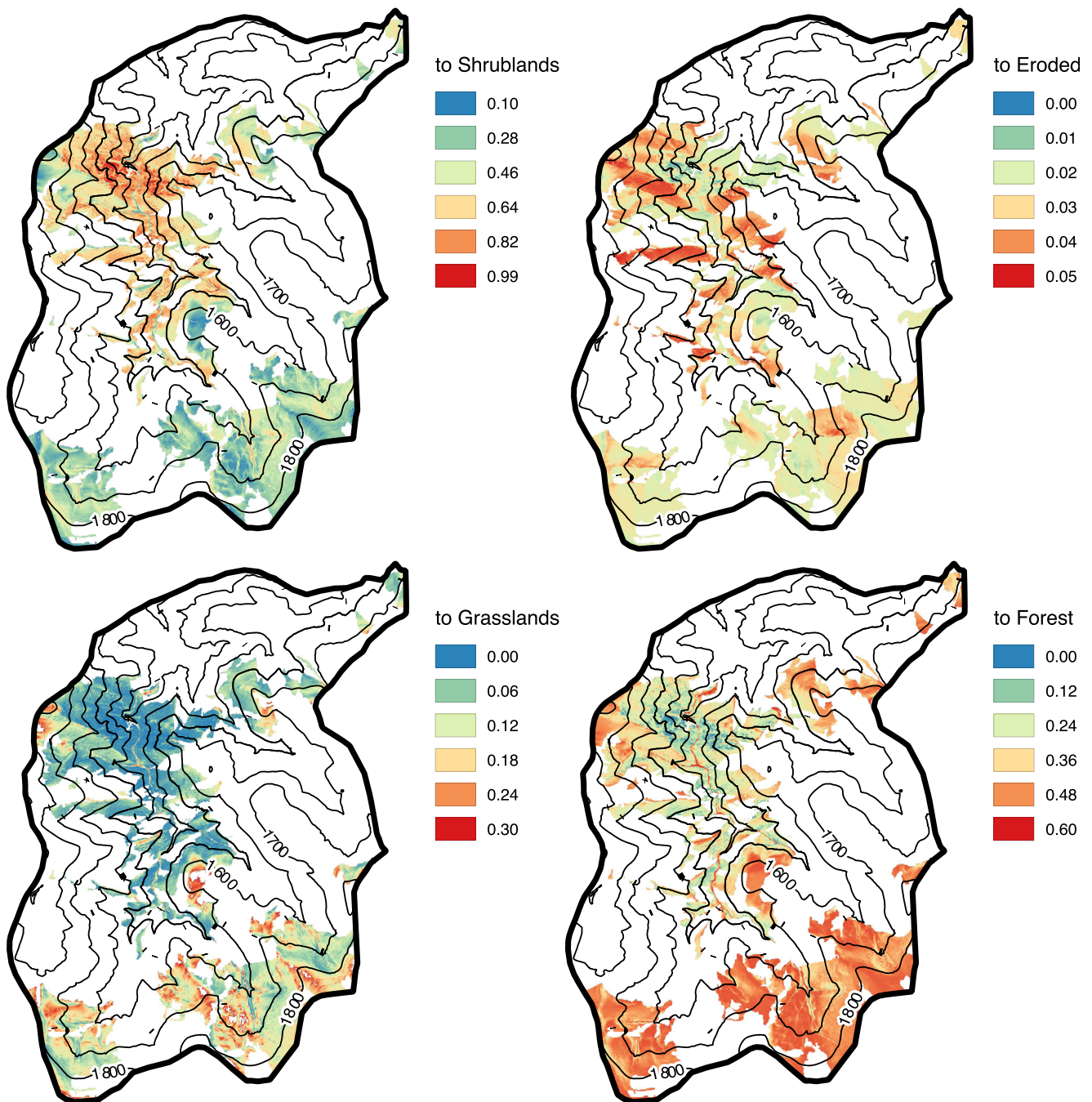


Fig. 6 Transition probabilities for the areas covered by shrubland in 1956: shrubland permanence and transition to forest, eroded areas, and grassland

of dense pine forest, a decline in the area of subalpine grasslands because of the advance of pine forest towards the upper part of the basin, and a decline in the area occupied by shrubs and clear pine forest. In the eastern French Pyrenees (Ariège Valley), the population and livestock decline after 1950 was followed by an increase in pine and the presence of *Juniperus* pollen in the peat bog of Orry de Théo (Galop et al. 2011). Summer grasslands have been colonized by heathland (*J. communis*, *C. vulgaris*, and *Cytisus scoparius*).

A remarkable trend towards encroachment has been also reported in the Alps, particularly between 1650 and 2450 m, where 10% of the new forest areas correspond to upward shifts in altitude, and 90% is attributable to densification of clear forests (Gehrig-Fasel et al. 2007). In the Unteralptal valley, the shrub cover increased by 18.7% and the shrub cover by 32.6%, resulting in an increase in resistance to soil erosion. Abandonment is also the main factor explaining the regeneration of forests in the Austrian Alps, particularly in the altitudinal range 1500–2150 m (Fondevilla et al. 2016).

Forest expansion will probably continue in the Urbión Mountains, because not all the potential forest areas in the montane and the subalpine belts have been occupied by forest since land abandonment. This expansion will occur in a haphazard manner, controlled by topographic factors. For example, our analyses demonstrated the importance of slope gradient, altitude, and solar irradiation, the latter depending on the slope gradient and aspect.

The presence of sheep flocks and cattle in summer could delay the advance of dense forest into the subalpine grasslands, whereas the advance will occur over several decades in the shrublands and clear forests. Forest expansion will not be possible into the eroded areas (caused by deforestation since the Neolithic period), so a complex mosaic of bare areas and summer grasslands will remain. These changes will have ecological, hydrological, and geomorphological consequences, including (i) increasing homogenization of the landscape, with larger patches dominated by dense forest surrounding shrinking summer grasslands and eroded areas; (ii) a decrease in the capacity of the massif for runoff generation, as has been demonstrated worldwide in mountains where dense forest has substituted grasslands and cultivated fields (García-Ruiz et al. 2011; García-Ruiz and Lana-Renault 2011), being this the main reason explaining the declining discharge of the rivers in Mediterranean mountains (Beguería et al. 2003; López-Moreno et al. 2014; Morán-Tejeda et al. 2014); (iii) a decline in soil erosion and shallow landslides, which in turn will result in reduced sediment supply to the streams as confirmed in Slovenian (Keesstra 2007), French Alps (Liébault and Piégay 2001; Piégay et al. 2004), and Pyrenean rivers (Beguería et al. 2006; Sanjuán et al. 2016); (iv) a reduction in the variety of functions and ecosystem services provided by summer grasslands, given the spatial contraction of grazing areas (Gartzia et al. 2014); and (v) possible changes in the fauna and floristic composition. The consequences of these landscape changes also include the timing of snowmelt in a landscape dominated by dense forest rather than shrublands and summer grasslands. Many uncertainties about the future of the upper montane and subalpine belts remain, and more research is needed to understand the changes that are occurring and their effects. This is the case of the future ecological, geomorphological, and hydrological implications of climate change and, above all, of a declining snow accumulation during the cold season because of global change (López-Moreno 2005; López-Moreno et al. 2009; Navarro-Serrano and López-Moreno 2017; Vicente-Serrano et al. 2017). This is particularly interesting in areas that are critical for water resources management at regional and national scales, and for sediment transport to the lowlands.

Concluding remarks

Deforestation of the subalpine belt in the Urbión Mountains occurred since Neolithic times, resulting in the transformation of forests into summer grasslands to feed the transhumant sheep flocks. This caused severe soil erosion because of shallow landsliding. Recent decline in the number of sheep flocks has resulted in changes in land cover reorganization, including the following:

1. The expansion of dense and clear forests at the expense of shrubs and grasslands.
2. The average 200 m altitudinal advance of dense forests.
3. The progressive spatial contraction of shrublands, mainly substituted by dense forest.
4. Forest expansion will continue in the next future since shrublands and grasslands are potential forest areas. Nevertheless, this trend will be conditioned by the evolution of the number of sheep and cattle flocks, because grazing controls woody encroachment.

The land use evolution detected in the subalpine belt of other Mediterranean mountains will result in land cover changes with highly relevant ecological, hydrological, and geomorphological consequences.

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