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The Role of Recent (1985–2014) Patterns of Land Abandonment and Environmental Factors in the Establishment and Growth of Secondary Forests in the Iberian Peninsula

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Abstract: Farmland abandonment has been a widespread land-use change in the Iberian Peninsula since the second half of the 20th century, leading to the establishment of secondary forests across the region. In this study, we aimed to address changes in the recent (1985–2014) emergence patterns of these forests and examine how environmental factors affected their growth by considering differences in leaf-habit types. We used a combination of Landsat-derived land-cover maps and aboveground biomass (AGB) maps from the European Space Agency to assess the secondary forest establishment and growth, respectively, in the study region. We also obtained a set of topographic, climatic and landscape variables from diverse GIS layers and used them for determining changes over time in the environmental drivers of forest establishment and AGB using general linear models. The results highlight that secondary forest cover was still increasing in the Iberian Peninsula at a rate above the European average. Yet, they also indicate a directional change in the emergence of secondary forests towards lower and less steep regions with higher water availability (mean rainfall and SPEI) and less forest cover but are subjected to greater drought events. In addition, these environmental factors differentially affect the growth of forests with different leaf-habit types: i.e., needleleaf secondary forests being less favoured by high temperature and precipitation, and broadleaf deciduous forests being most negatively affected by drought. Finally, these spatial patterns of forest emergence and the contrasting responses of forest leaf-habits to environmental factors explained the major development of broadleaf evergreen compared to broadleaf deciduous forests and, especially, needleleaf secondary forests. These results will improve the knowledge of forest dynamics that have occurred in the Iberian Peninsula in recent decades and provide an essential tool for understanding the potential effects of climate warming on secondary forest growth.

Keywords: drought; forest expansion; land-cover change; land-use change; rural abandonment

1. Introduction

Changes in land use and land cover are one of the main drivers of global environmental change given that they affect the properties of land surfaces, the provision of ecosystem services [1] and, ultimately, the Earth's energy balance and biogeochemical cycles [2,3]. Although deforestation is still one of the main manifestations of changes in land use and land cover, the net rate of forest loss decreased substantially worldwide during 1990–2020

due to a reduction in deforestation in certain countries and active and passive afforestation in others [4]. The expansion of secondary forests, i.e., forests that are established in areas with another previous land use, is a phenomenon that has occurred especially in the Northern Hemisphere during the last few centuries [5]. Secondary forests that have been established on former farmland may constitute a large proportion of the total forested area in many regions, e.g., 20–25% in Spain [6,7] and more than 50% in China and Costa Rica [8].

The expansion of secondary forests can have both positive and negative effects on landscape structure and ecological functions. For example, forest cover increase is affecting biodiversity conservation in Europe, with a generalised recovery of forest species, including threatened species targeted in conservation initiatives [9,10], but also a rarefaction and local extinction of species that are typical of open habitats [10–12]. The expansion of secondary forests may also increase the functional connectivity between forests (e.g., seed dispersal potential) and may facilitate migration and gene flow between tree populations in response to climate change [13], while potentially increasing the propagation of large wildfires, especially in southern Europe [14]. Apart from these environmental changes, the expansion of secondary forests has gained much attention owing to their role in carbon accumulation [15,16]. Fuchs et al. [17] determined that afforestation and cropland abandonment made the greatest contribution to carbon sequestration during 1950–2010 in Europe, while Pan et al. [16] reported that the C sink of China's forests increased by 34% during 1990–2007 primarily due to newly planted forests. Finally, Vila-Cabrera et al. [6] showed that secondary forests that were established from 1956 onwards represent 22% of the total C pool in Iberian forests. Furthermore, it was suggested that forests established on former agricultural land may differ from long-established forests in terms of species composition and their structural and functional characteristics [18]. These forests may benefit from land-use legacies since their soils tend to be richer in nutrients [19] and exhibit greater enzymatic activity [20], which could explain why some secondary forests have higher growth rates than long-established forests: 35% greater plant biomass [21] and 25% higher growth [6], but they may also be more sensitive to climate changes due to differences in functional attributes (lower wood density [22] and finer root morphology [21,23]). This may be of special relevance under climate change, as drought and heat-induced stress in trees could lead to a reduction in tree growth [24], an increase in mortality rates [25,26] and an increase in forest vulnerability [27,28]. Ultimately, this may cause directional changes in forest composition towards forests with a greater abundance of drought-tolerant species [29,30], e.g., an increase in broadleaf over needleleaf forests [31,32]. On the other hand, the spatial pattern of forest expansion is mostly determined by farmland abandonment, which is supposed to be concentrated in marginal and low productivity areas [33]. Ultimately, the biophysical conditions involving forest expansion may influence the recruitment, growth and mortality of tree species affecting the whole plant community [34–36], which may determine secondary forest productivity and its capacity for carbon accumulation [15,16,34,37]. Although some studies have suggested that the expansion of secondary forests may contribute in a relevant way to carbon sequestration [6,15–17], little is known about the influence of the expansion patterns on the extent of secondary forest growth.

The purpose of this study was thus to determine the recent patterns (1985–2014) of secondary forest establishment and growth in the Iberian Peninsula and examine how environmental factors affect them by considering the main forest leaf-habit types. The Iberian Peninsula is an especially appropriate study area given, on the one hand, the expansion of secondary forests that occurred there during the second half of the twentieth century [6,7,33] and, on the other, its diverse topoclimatic conditions and the increasing number of drought episodes in recent decades [38]. Ultimately, this may help to evaluate the potential of recently established forests to climate change mitigation strategies that are aimed at maximising carbon sequestration.

We thus established three main hypotheses: (i) the distribution and magnitude of cropland abandonment are closely determined by the environmental characteristics that limit their productivity and, as such, secondary forests will appear first in poorly productive

areas (i.e., areas with unfavourable climatic and topographic conditions) since the most productive areas will be the last to be abandoned; (ii) due to the recent increase in drought frequency and severity in recent decades, a greater amount of drought-tolerant broadleaf secondary forests than needleleaf forests will develop; and (iii) drought events will have a greater impact on needleleaf secondary forest growth than on broadleaf secondary forest growth.

2. Material and Methods

2.1. Study Area

The study was performed in the Iberian Peninsula (202,067 km^2; SW Europe), a region with great climatic and topographic diversity (Figure 1) and three biogeographic regions, namely, Mediterranean, Atlantic and Alpine. The mean annual temperatures range from 18 °C (on the southern coast) to 1 °C (in mountainous areas), while the mean annual rainfall varies from 340 mm to over 2400 mm (Topoclimatic Drought Atlas of the Spanish Iberian Peninsula; Domingo-Marimon, 2016) [39]. The large number of mountain ranges (from sea level to more than 2600 m a.s.l.) and steep coastal-inland gradients also contribute to the great environmental heterogeneity of this region. Forests cover 35% of the Iberian Peninsula (EEA Report No. 5/2016). Broadleaf and needleleaf evergreen species are dominant in the study area; broadleaf deciduous forests are less frequent and are mostly found in the Atlantic region and mountainous areas (see Figure 1). The main needleleaf evergreen species in Mediterranean areas are *Pinus halepensis* and *P. pinea*, while *P. nigra* occurs in inland upland areas. *Abies alba* and *P. uncinata* are common in Alpine areas in the Pyrenees, while *P. sylvestris* is found widely from upland Mediterranean to Alpine areas. Broadleaf evergreen forests are dominated by *Quercus ilex* and *Q. suber* in lowland sites up to the limit of montane habitats, while *Q. coccifera* becomes increasingly common as continentality rises. Broadleaf deciduous forests mainly consist of *Q. humilis*, *Q. faginea* and *Q. pyrenaica* in Mediterranean lowlands and uplands, and *Fagus sylvatica* in the Atlantic region.

2.2. Data Sources

To detect the secondary forest emergence, we used Landsat images that were processed and classified by the Grumets Research Group at the Autonomous University of Barcelona (www.ogc3.uab.cat/acapi/wms/USOS/index.htm; accessed on 10 November 2020, [40,41]). We ensured that the maps were only generated for those years with enough good-quality data and periodicity (i.e., from the mid-1980s). In order to have enough images for map classification (i.e., images that enabled us to correctly identify phenological variation in land cover categories), we worked with quinquennial periods, thus generating maps for every quinquennia from 1985 to 2014. From these maps, we selected a set of available scenes from the Landsat orbits (197, 198, 200, 201 and 202) that were distributed along both latitudinal and longitudinal gradients that were representative of the climatic variability in the Iberian Peninsula (Figure 1).

Secondary forest growth was estimated by using the global maps of aboveground biomass from 2017 (100 m resolution) that were generated by the European Space Agency (ESA) Climate Change Initiative Biomass project [42]. Aboveground biomass (AGB) is defined as the amount of living biomass (organic matter) that is stored in vegetation above the soil, including stems, stumps, branches, bark, seeds and foliage, and is expressed as the weight of dry matter per unit ground area (i.e., Mg/ha^{-1}) [43].
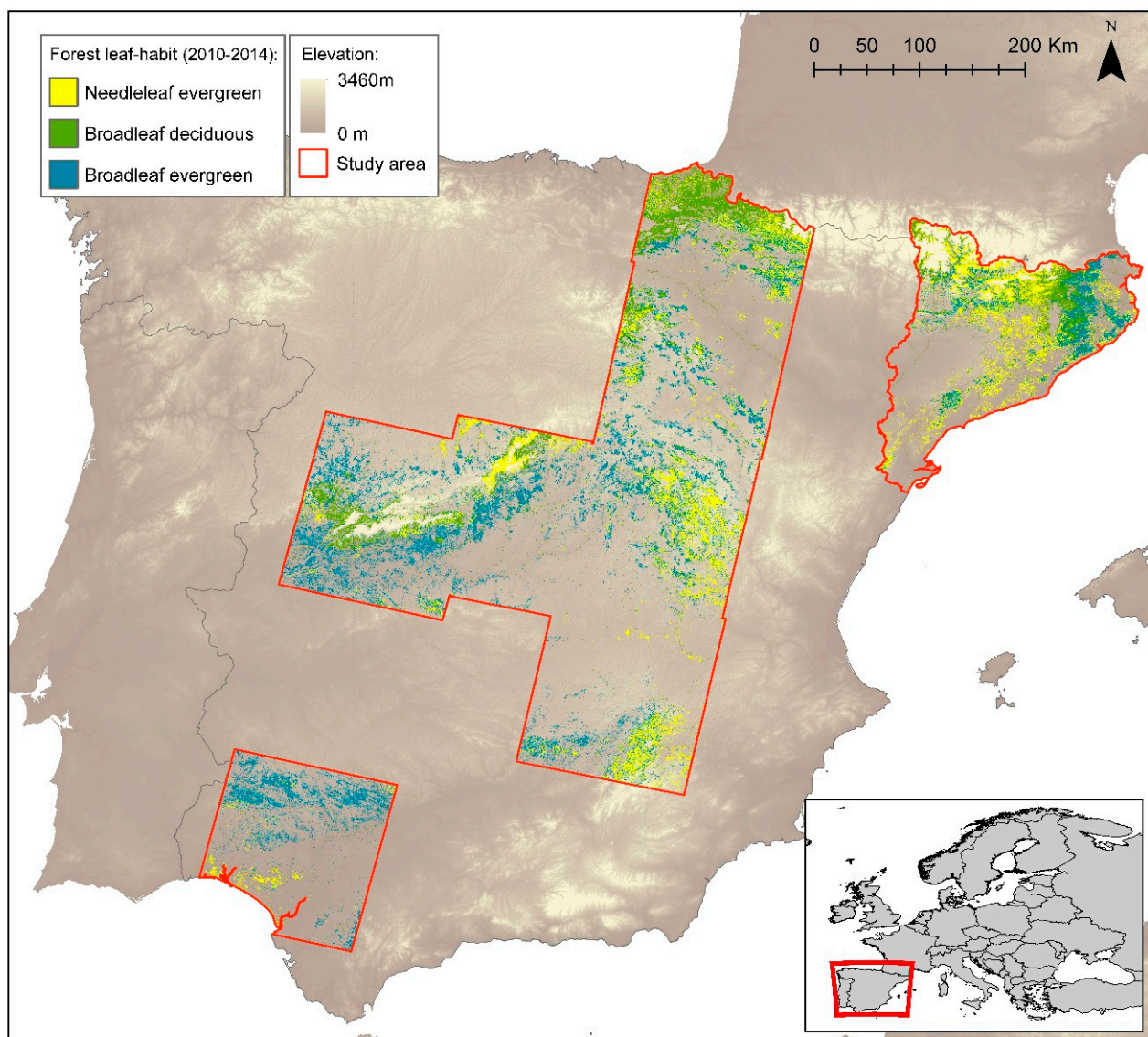
Figure 1. Secondary forest distribution during 2010–2014 by the main forest leaf-habit types in the Iberian Peninsula.

2.3. Detecting Secondary Forest Establishment and Growth throughout the Study Period

We detected a new secondary forest that had appeared in the study period by combining maps relating to the different quinquennia. We defined a secondary forest as a forest that was established on cropland or grassland and is a product of forest expansion, that is, forests established in areas that for many years previously had been dominated by a different type of land use [44]. We did not consider the transition from shrubland to forest as a secondary forest since this was explored by previous studies [45] and because this transition mostly corresponds to the regeneration of previously felled or disturbed forests [7,46].

For each quinquennium, we identified pixels belonging to forest land cover that was cropland or grassland in the previous quinquennium (see classes in the Online Resource 1, Table S1). To avoid including pixels that had been falsely classified as new forests, we only selected forests in a quinquennium that appeared as forests in two subsequent quinquennia (i.e., cropland or grassland during 1995–1999 that was classified as forest in both 2000–2004 and 2005–2009). To account for potential differences that were related to leaf-habit types, we determined surface changes and annual rates of change for the main leaf habit types: (i) broadleaf deciduous, (ii) broadleaf evergreen and (iii) needleleaf evergreen.

Finally, the growth of the selected secondary forests was assessed by estimating their biomass in relation to their age. Biomass was obtained by merging the detected secondary forests during 2010–2014 with the ESA aboveground biomass (AGB) layer. We previously resampled the secondary forest dataset at a pixel size of 100 m (using the modal criterion of the most represented value) to fit with the resolution of the biomass dataset. Forest age was assessed in years as the difference between the final quinquennium (2010–2014) and the quinquennium of forest establishment.

2.4. Environmental Drivers of Secondary Forest Establishment and Growth

To assess the environmental context of the secondary forests in the study area, we selected a set of potential drivers of forest expansion and growth [45,47–50]. Based on previous work (e.g., [45,50–52]), we chose a series of climatic, topographic and landscape factors that are known to affect forest composition, structure and growth.

2.4.1. Climatic Variables

To characterise the main climatic conditions, we used the Topoclimatic Drought Atlas of the Spanish Iberian Peninsula [39], which includes monthly aggregates from 1950–2015 at a spatial resolution of 100 m of rainfall, mean temperature and the Standardised Precipitation Evapotranspiration Index (SPEI) as an indicator of water deficit. We calculated the mean temperature (°C) and rainfall (mm) annually and seasonally for 1950–2015 (hereafter, the historic climate) and for the time that the secondary forest grew (hereafter, the recent climate). We used SPEI values that were calculated at a timescale of 12 months, which is an optimal and appropriate scale for studying the long-lasting dry periods that characterise hydrological droughts [53,54]. We defined a drought event as a period of consecutive months in which the SPEI values were equal to or lower than −1, as suggested in the literature [53,55,56]. For a statistical quantification of drought episodes, we calculated the commonly used drought parameters [55,57] for both the historic and recent climatic periods, i.e., drought frequency (number of episodes during a period divided by period; events/year), mean drought duration (average duration of drought events in the period; months), mean drought intensity (average of the mean SPEI values during the period of drought events) and mean drought severity (average of the accumulated SPEI values during the period of drought events).

2.4.2. Topography

We used the digital elevation model of the Iberian Peninsula at a spatial resolution of 90 m generated by the Kraken group from the University of Extremadura [58] to characterise the mean elevations (m) and slopes (degree) of emerging secondary forests.

2.4.3. Forest Cover

Previous studies have observed that the amount of forest cover in the landscape has a positive influence on forest expansion and determines the distribution patterns of secondary forests [45,50]. Thus, we determined the percentage of forest land cover at the beginning of the study period by counting the number of forest pixels in a radius of 1 km of each secondary forest pixel. Finally, we resampled the spatial resolution of the abovementioned land-cover raster datasets from 30 m to 100 m to coincide with the resolution that is employed in most environmental datasets generated using the majority criterion (assigning the value of each pixel based on the most abundant value). We overlapped multiple environmental maps, after resampling and changing the coordinate systems wherever necessary, to determine the environmental values in all cells classified as secondary forests.

2.5. Statistical Analyses

As for the secondary forest establishment, we first performed ANOVA tests to determine whether the environmental conditions of established secondary forests differed between forest leaf-habit types (broadleaf deciduous, broadleaf evergreen and needleleaf

evergreen). Data were log-transformed when necessary to meet the assumptions of normality. Where significant differences occurred ($p < 0.05$), comparisons between means were performed using Tukey's HSD multiple comparison test.

We then performed a general linear model (GLM) to test the association of the time (quinquennium) of the establishment of secondary forests with the environmental variables described above. For this analysis, we took the historic climatic variables from the period 1950–2015 as being representative of the climatic conditions in the area. We decided to work with the absolute value of the severity and intensity of drought factors (all values were negative) so that the higher the value, the greater the severity or intensity of the drought, which facilitated the interpretation of the effect in the model. To prevent multicollinearity, we produced a correlation matrix with a Spearman rank test (Electronic Supplementary Materials 2, Figure S1) to avoid using highly correlated variables ($r < |0.7|$). Thus, when variables were correlated, we chose the best variable by selecting the one with the strongest effect on the response variable. The simplest general linear models were selected following a dredge procedure using the MuMIn R package [59], which removed non-significant variables from the general model, to assess significant changes in model predictions using the Bayesian information criterion (BIC). From the models with a difference in BIC relative to BICmin < 2, we chose the most parsimonious model by selecting the model with the fewest predictor variables following the procedure described in Crawley [60]. Then, we tested for spatial autocorrelation of the model residuals using Moran's I Autocorrelation Index [61] to ensure sampling independence (Electronic Supplementary Materials 1, Table S2; [62]).

As for forest growth, we first performed ANOVA tests for each forest quinquennium to determine whether AGB differed between forest leaf-habit types and performed pair-wise comparisons of means using Tukey's HSD tests wherever significant differences occurred ($p < 0.05$).

We also performed a GLM to test the association of AGB with environmental variables and age (midpoint of each quinquennium). For this analysis, we used the recent climatic variables calculated from the year the secondary forest became established to the end of the study period (2015) to describe the climatic conditions in which secondary forests grew. In this case, we also used absolute values of the drought severity and intensity factors. We employed the same criteria to prevent multicollinearity and avoid highly correlated variables used in the abovementioned model (Electronic Supplementary Materials 2, Figure S2). Furthermore, we considered forest leaf-habit as a factor and the environmental interactions with the forest leaf-habit. The simplest general linear model was selected using the same criterion as described above. Then, as for the model analysing forest establishment, we tested for spatial autocorrelation of the model residuals using Moran's I Autocorrelation Index (Electronic Supplementary Materials 1, Table S3).

3. Results

3.1. Secondary Forest Establishment

The total forest area (i.e., the balance between gains and losses) increased by 7.7% during 1985–2014, which represents a mean annual increase of 0.31%. Broadleaf deciduous (BD) forests showed the highest increase from their initial area (30.3% during the study period, 1.21% annually). Broadleaf evergreen (BE) forests increased by 5.2% (annual increase 0.21%), while needleleaf evergreen (NE) forests showed no changes in their area.

Secondary forest establishment occurred at an annual rate of $0.60 \pm 0.15\%$ (mean ± SE of quinquennia). BD forests had the highest annual establishment rate ($1.09 \pm 0.21\%$), followed by BE ($0.81 \pm 0.32\%$) and NE ($0.24 \pm 0.11\%$) forests. By the end of the study period, 14.8% of the forest area corresponded to secondary forests that had emerged during the 1985–2014 period (54.2% of BD, 31.1% of BE, and 14.7% of NE forests; see Figure 2 for the spatial distribution of secondary forests by forest leaf-habit).
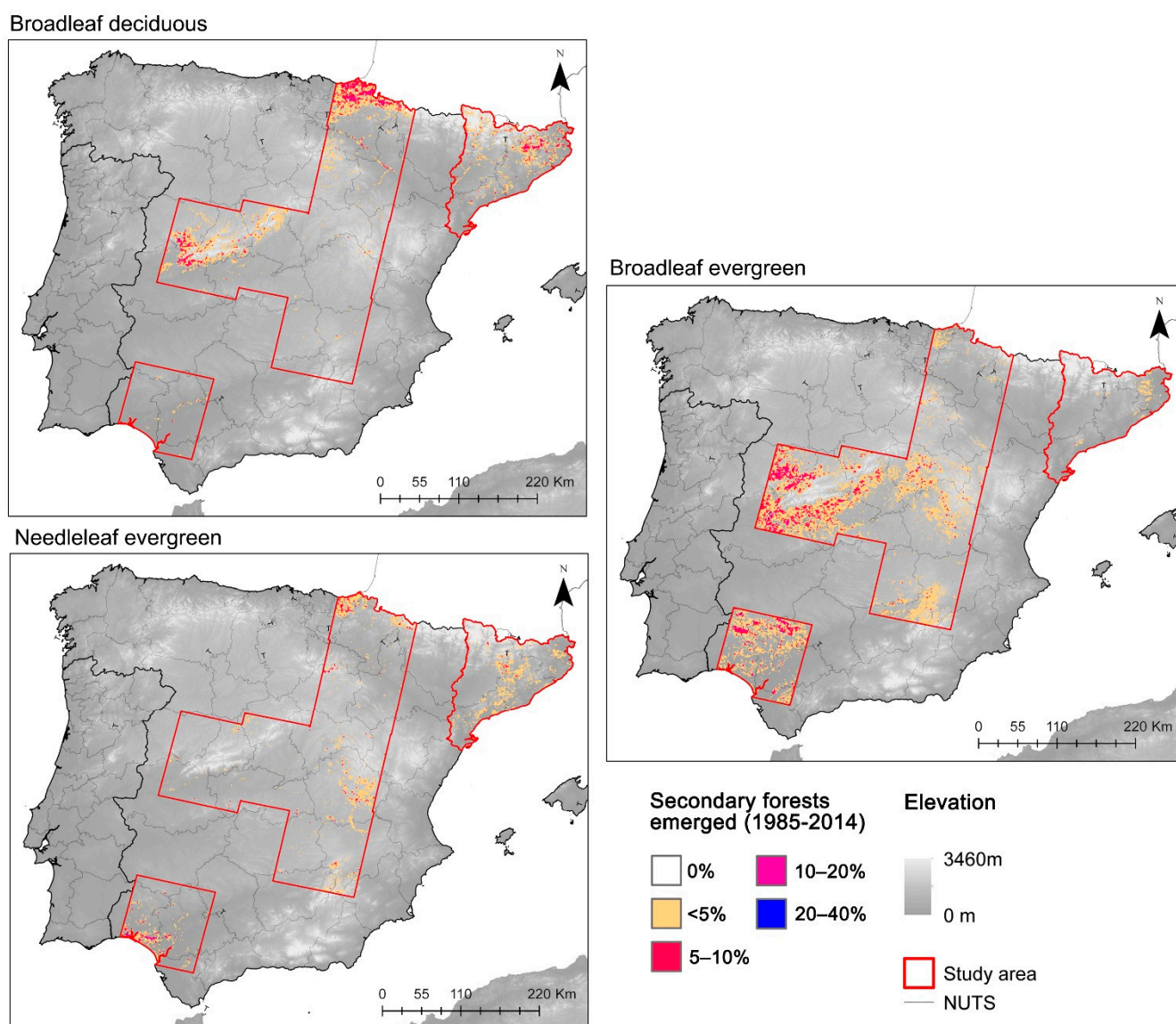
Figure 2. Secondary forest distribution and area (percentage of 10 km pixels) during 1985–2014 for the main forest leaf-habit types in the Iberian Peninsula: broadleaf deciduous, broadleaf evergreen and needleleaf evergreen.

Secondary forest establishment occurred in areas with a mean ($\pm$ SE) elevation of 667 $\pm$ 0.8 m, a slope of 6.2 $\pm$ 0.0% and a previous tree cover of 30.3 $\pm$ 0.05%. The mean climatic conditions of these areas were: annual temperature: 13.6 $\pm$ 0.01 °C, annual precipitation: 710 $\pm$ 0.49 mm, 0.46 $\pm$ 0.00 droughts/year with 4.74 $\pm$ 0.00 months of duration, intensity of SPEI values: -1.30 ± 0.00 and severity of SPEI values: -6.91 ± 0.00. BD secondary forests became established in areas with higher average annual precipitation rates, steeper slopes and more forest cover but with a lower mean annual temperature than BE and NE forests (Table 1). They also became established in areas with lower elevations, on less steep slopes and lower average annual precipitation, but with higher mean annual temperatures and greater drought intensity and severity than BD and NE forests. NE forests became established in areas of greater environmental variability with less drought duration, intensity and severity than either BD or BE forests.

Table 1. Mean (±SE) values of environmental variables for sites with secondary forests and differences between secondary forest leaf-habit: broadleaf deciduous (BD), broadleaf evergreen (BE) and needleleaf evergreen (NE). Letters show significantly different means at $p < 0.05$.

	Secondary Forest Environment					
	Elevation (m)		**Slope (%)**		**Forest cover (%)**	
All	**667 ± 0.8**		**6.17 ± 0.01**		**30.3 ± 0.05**	
BD	681 ± 1.5	a	8.26 ± 0.02	a	36.7 ± 0.10	a
BE	666 ± 1.1	b	5.25 ± 0.02	b	27.6 ± 0.06	b
NE	684 ± 1.8	a	6.05 ± 0.03	c	29.2 ± 0.11	b
	Mean annual temperature (°C)		**Annual precipitation (mm)**		**Drought frequency (event/year)**	
All	**13.6 ± 0.01**		**710 ± 0.49**		**0.46 ± 0.00**	
BD	12.5 ± 0.01	a	895 ± 0.87	a	0.46 ± 0.00	a
BE	14.0 ± 0.01	b	654 ± 0.59	b	0.46 ± 0.00	a
NE	13.7 ± 0.01	c	623 ± 1.00	c	0.48 ± 0.00	b
	Drought duration (months)		**Drought severity**		**Drought intensity**	
All	**4.74 ± 0.00**		**−6.91 ± 0.00**		**−1.30 ± 0.00**	
BD	4.72 ± 0.00	a	−6.88 ± 0.00	a	−1.29 ± 0.00	a
BE	4.79 ± 0.00	b	−6.98 ± 0.00	b	−1.31 ± 0.00	b
NE	4.60 ± 0.00	c	−6.74 ± 0.00	c	−1.30 ± 0.00	c
	SPEI		**Longitude**		**Latitude**	
All	**0.004 ± 0.0000**		**435,901 ± 440**		**4,453,824 ± 369**	
BD	0.005 ± 0.000	a	552,225 ± 786	a	4,604,454 ± 642	a
BE	0.005 ± 0.000	b	349,954 ± 535	b	4,401,588 ± 436	b
NE	0.004 ± 0.000	c	526,596 ± 899	c	4,404,518 ± 733	c

The best model describing the environmental factors that were associated with the time of establishment of the secondary forests during 1985–2014 is shown in Figure 3. The time of establishment of the secondary forests (quinquennial) was positively associated with the mean summer precipitation, mean SPEI and mean drought intensity, but negatively associated with slope and forest cover. Significant interactions of forest leaf-habit with some environmental factors in the model indicated that, over time, both BE and NE were established in areas with lower elevation, lower forest cover areas and lower drought intensity compared with BD forests, while BE forests were established in less steep areas than both NE and BD forests. An autocorrelation test (Electronic Supplementary Materials 1, Table S2) detected very low yet significant values of the Moran's Index on the study sample and its model residuals (0.071 and 0.027, respectively).

3.2. Secondary Forest Growth

The aboveground biomass (AGB) of the secondary BD forests had mean values (±SE) ranging from 61.3 ± 1.1 Mg ha^{-1} in the youngest forests (5 years) to 101.5 ± 0.9 Mg ha^{-1} in the oldest (25 years). These values were higher than those of BE and NE forests for most age groups (Figure 4). By contrast, secondary BE forests had the lowest AGB values for most age groups, ranging from 35.0 ± 0.7 Mg ha^{-1} to 58.3 ± 0.6 Mg ha^{-1}. Finally, NE secondary forests had mean AGB values ranging from 37.3 ± 1.4 Mg ha^{-1} to 77.9 ± 0.9 Mg ha^{-1}.

The GLM model for the effect of local environmental factors on the growth of secondary forests explained 42% of the variance (Figure 5). The growth of secondary forests was positively associated with age, slope, forest cover, summer precipitation and mean annual temperature, but negatively associated with drought frequency and intensity. Additionally, BE forests had the lowest AGB, while NE forests had the highest AGB. In addition, the GLM revealed significant interactions between the forest leaf-habit and climatic factors,

that is, the positive effect of summer precipitation was higher for BE forests than for NE and BD forests, the positive effect of temperature was lower in NE forests than in BE and BD forests, while the negative effect of drought frequency and intensity was lower in BE and NE forests than in BD forests. Finally, both slope and summer precipitation had a negative interaction, which indicates that the greater the slope, the lower the positive effect of summer precipitation. The autocorrelation test (Electronic Supplementary Materials 1, Table S3) showed large decreases in the Moran's I values from the study sample (0.252) to the residuals of the GLM model with the explanatory variables (0.005), with both values being significant.
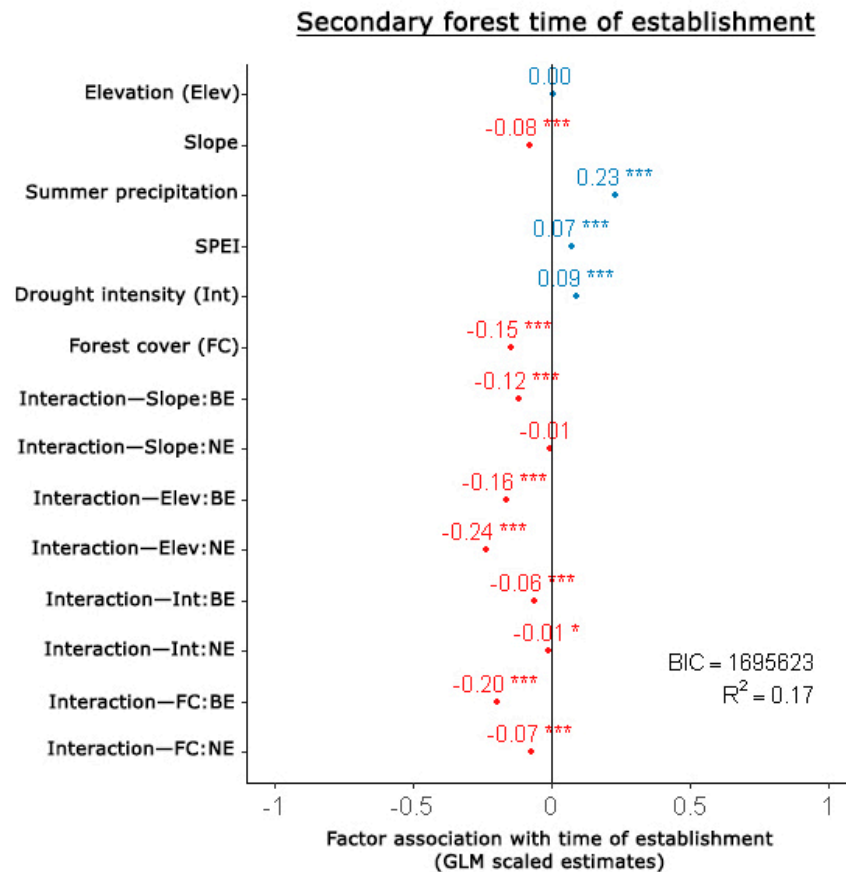

Figure 3. General linear model (GLM) parameter estimates of the relationships between the time (quinquennium) of secondary forest establishment and elevation (m), slope (°), annual precipitation (mm), mean annual SPEI values, drought intensity, forest cover in a 1 km radius (%) and interaction with secondary forest leaf-habit broadleaf evergreen (BE) and needleleaf evergreen (NE) forests compared to broadleaf deciduous (BD) forests. Climatic values were calculated from the mean of the period 1950–2015. Significance is indicated using the following *p*-values: *: 0.05, ***: 0.001.
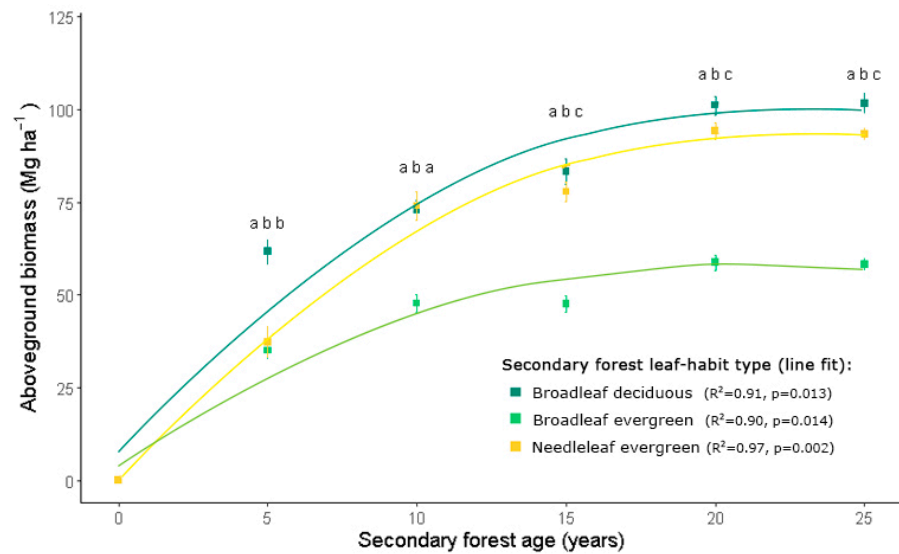
Figure 4. Average aboveground biomass (AGB) of secondary forests by age for the different forest leaf-habit types. Different letters indicate significant differences in AGB between forest leaf-habit types for the same age according to Tukey's test ($p < 0.05$).
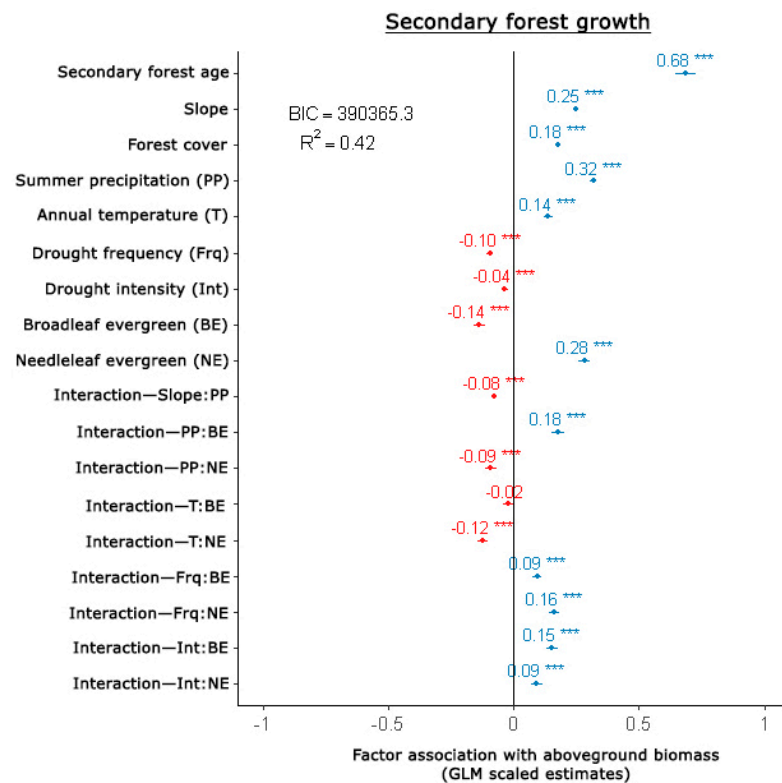

Figure 5. General linear model (GLM) parameter estimates of the relationships between secondary forest growth (aboveground biomass; Mg ha^{-1}) and several explanatory variables, including the logarithm of the secondary forest age (y), slope (°), forest cover in a 1 km radius (%), mean precipitation in summer (mm), mean annual temperature (°C), drought frequency (events/year), drought intensity and forest leaf-habit (broadleaf evergreen (BE) and needleleaf evergreen (NE) compared to broadleaf deciduous (BD)). Climatic factors were calculated using the mean of the secondary forest establishment periods (from the quinquennium of establishment up to 2015). Significance is indicated using the following *p*-values: ***: 0.001.

4. Discussion

4.1. Forest Expansion in the Iberian Peninsula

Results show that forest area in the Iberian Peninsula is increasing at an average net rate of 0.31% year^{-1}, which is a rate that is much higher than the European average in recent decades (0.08% year^{-1} during 1990–2015 [63] and 0.06% year^{-1} during 1992–2015 [45]) and closer to the European rate for the twentieth century (0.50% year^{-1} during 1950–2000 [64] and 0.42% year^{-1} during 1900–2010 [65]). This reflects the current great forest dynamism in the Iberian Peninsula [66], which may continue in the near future [67], compared to other regions of Europe where forest cover is stabilising [4,64,68]. This phenomenon can be attributed to the massive cropland abandonment that occurred in southern (and eastern) Europe during the application of the Common Agricultural Policy (CAP) during 1988–2008, which forced agriculture to be more competitive in global markets [33,64,69,70] and prompted the afforestation of former croplands (e.g., the program *Programa de Forestación de Tierras Agrarias* (FTA) in Spain [71]).

The gross rate of secondary forest establishment determined in this study (0.60% year^{-1}) during 1985–2014 is generally consistent with the results reported by Vilà-Cabrera et al. [6] for 1956–2007 (0.75–0.43% year^{-1}). Moreover, our results show that the secondary forest establishment rate differed according to the main forest leaf-habit types: NE had the lowest rate (0.24 % year^{-1}) and represented only 14.7% of the total secondary forests, while BD and BE had higher rates (respectively, 1.21% and 0.81% year^{-1}) and represented 31.1% and the 54.2% of total secondary forests, respectively. This may be attributed to the prioritisation of broadleaf species (i.e., *Quercus* spp.) in the FTA program: monospecific stands of broadleaf species have represented 50% of the total afforested area in Spain since 1992, while stands of needleleaf species have represented only 15% [71]. Nevertheless, it could also be due to the different responses of broadleaf and needleleaf species to the biophysical attributes of abandoned land, which is a consideration that is discussed in more detail below.

4.2. Patterns of Secondary Forest Establishment

According to our first hypothesis, the establishment of secondary forests is increasing over time in more favourable biophysical conditions, as shown by the GLM model. There is a directional change over time of secondary forest emergence towards lower elevation and less steep regions with higher water availability (i.e., higher average summer precipitation and annual SPEI values; Figure 3). However, directional change in areas with greater drought intensity was also detected. These changes over time suggest that, while global or external causes may trigger the abandonment of croplands and pastures, local or regional factors constrain the degree and location of the abandonment [5,33]. Thus, in those regions where crop and pasture abandonment occur, the first areas to be abandoned are the sites of least quality (e.g., in terms of climate limitations and slope steepness) that limit productivity and hamper the mechanisation of tasks. Conversely, the directional change of forest expansion towards regions with a higher average drought intensity is difficult to interpret due to the heterogeneous spatiotemporal patterns of drought occurrence in the Iberian Peninsula [72]. Furthermore, although the average annual temperature and precipitation determine climatic conditions and thus the potential agricultural productivity and land abandonment [5,33], drought events may not have this effect because of their episodic occurrence over both space and time [73]. In any case, the positive relationship with drought intensity could be a collateral effect of the directional trends of forest emergence towards lower elevation areas with higher annual temperatures, given that temperature plays a key role in drought intensity in the Iberian Peninsula [74].

In addition, the effects of topographic factors interact with forest leaf-habit, as indicated above (Figure 3). A decrease in forest establishment over time with elevation was detected in BE and NE forests but not in BD forests. We attribute this to the fact that the BD forest distribution in the Iberian Peninsula is mostly constrained to mountain ranges with higher precipitation rates than surrounding lowland areas enjoying a typical Mediterranean climate [75]. By contrast, NE secondary forest emergence decreased most with elevation

over time, probably because this forest leaf-habit has the widest altitudinal distribution, ranging from the *Abies alba* and *P. uncinata* stands at high elevations to the *P. halepensis* forests that dominate in lowlands. On the other hand, the directional change of secondary forest emergence towards less steep areas was far more evident in BE forests than in NE and BD forests. This may be due to the fact that a large proportion of BE forests (i.e., those dominated by *Quercus ilex*) were historically restricted to non-cultivated steep areas in mountain ranges where they were exploited for charcoal production [76], while NE and BD forests often persisted while intermingled with croplands and pastures in the plains. Moreover, as these BE forests are distributed from near the coast to the upper montane limit, their establishment pattern may be better described by slope than by elevation.

We also detected that the time of establishment of secondary forests was negatively related to forest cover in the surrounding area, probably due to the change in forest emergence towards lowland, less steep areas. This may also explain why the negative effect of forest cover on secondary forest emergence was more negative in BE and NE forests than in BD secondary forests, given that the latter are mostly found in northern and upland areas in the Iberian Peninsula (corresponding to a temperate biome). In previous studies, we detected a positive relationship between previous forest cover percentage and forest expansion [45], which we attributed to the concentration of land abandonment in landscapes that were dominated by agroforestry mosaics [45,63]. In addition, reforestation is more likely to occur in regions with low forest cover and poor land suitability for agriculture but good connections to wood markets [5].

4.3. Secondary Forest Growth

The GLM assessing the effects of the diverse factors on forest growth explains 42% of the variability of secondary forest growth (Figure 5). As expected, age had the greatest effect on forest growth, although environmental and forest leaf-habit factors also had important implications for forest growth.

Summer precipitation seems to be a key element in forest growth in the study area [77], which is very dependent on the cumulative spring and summer water deficit [78]. This also depends on the forest leaf-habit, with the effect being higher in broadleaf forests, especially evergreen forests, than in needleleaf forests. Our results corroborate the finding that under warm climatic conditions, broadleaf species tend to be much more competitive than needle-leaf species if soil moisture is available [79], as their roots are able to penetrate into the deep water table [80]. In addition, secondary forest growth was also positively related to mean annual temperature, with this effect being greater in broadleaf forests than in needleleaf forests. This agrees with previous studies [81,82] and can be attributed to the contrasting water economy strategies in broadleaf and needleleaf species [83]: broadleaf species, mainly oak species, have less strict stomatal control, which allows them to assimilate carbon for longer during warmer and drier periods; on the other hand, needleleaf species, i.e., pine species, typically have more isohydric behaviour and reduce their stomatal conductance to a minimum during the warm and dry seasons [84,85].

Our best model also revealed the negative effects of the drought frequency and intensity parameters on secondary forest growth. The magnitude of this effect depends on the forest leaf-habit, where the response to drought in BD secondary forests is more pronounced than in BE and NE forests. This may be caused by the fact that BD secondary forests in the study area are mainly found in the north of the Iberian Peninsula and consist in part of species at the southernmost limits of their ranges (e.g., *Fagus sylvatica* and *Quercus humilis*) that may have a lower buffering capacity than drought-tolerant species [54]. Moreover, recent studies suggest that beech forests established on former agricultural land have higher growth rates due to better soil attributes (i.e., higher N and P content and mineralisation rates), which also implies lower wood density [22,23]. This may also mean that BD secondary forests may be particularly vulnerable to drought-induced cavitation [86]. On the other hand, our model indicates that the growth of BE secondary forests was the least affected by drought frequency, while the growth of NE secondary

forests was the least affected by drought intensity. Once again, variations in water-use strategies between the different leaf-habits may be key for understanding the response of ecosystems to average climate and drought episodes [83]. Needleleaf species typically avoid drought by drastically reducing stomatal conductance at the first sign of water deficit [87,88], which implies that their growth is affected by drought frequency [25,78] but not by intensity given that they will stop growing when water deficit is present, thereby avoiding the adverse effects of intensity. Conversely, BE species usually maintain higher stomatal conductance, even at low leaf water potentials [87,88], which may explain the higher effect of drought intensity on BE secondary forest growth.

In addition, our results suggest that slope had a strong effect on secondary forest growth. Different effects of slope on forest growth have been reported [82,89–92] and attributed to associated biophysical properties (i.e., soil depth, solar radiation and mean elevation). Our results indicate a positive effect of slope on secondary forest growth, thereby suggesting that there was more growth in upland areas than in lowland plains. This may be because land abandonment was mostly prompted by low productivity (i.e., climatic limitation or soil degradation) in the plains and by topography (i.e., difficult access and a constraint on mechanisation), and not necessarily by productivity in upland areas [33,93]. Our findings also reveal that the positive effect of summer precipitation on growth decreases as slope gradients increase, which suggests that forests in plains may grow under arid and semiarid conditions [94] and, thus, that forest growth was most dependent on summer precipitation. Finally, the model shows that growth was higher in needleleaf forests than broadleaf secondary forests, which was probably due to the lower wood density and faster growth rates in the former forest type [81,95].

These results highlight that the spatial patterns of recent forest expansion in the Iberian Peninsula may be compromising needleleaf growth and establishment and favouring the establishment of broadleaf forests, as the latter may cope better with warm temperatures, benefit more from summer precipitation and its dispersal mediated by vertebrates, which facilitates the colonisation of less forested landscapes [31,48,96]. This may have advantages for forest functioning under the context of climate change, as broadleaf evergreen species have a more stable production in wet and dry periods [30] and are highly resilient to droughts [83] and wildfires due to their ability to reshoot [97,98]. However, this may entail that there will also be a global reduction in the rate of carbon fixation by secondary forests in the Iberian Peninsula, as forests that are more tolerant to drought are less productive [30,99].

5. Conclusions and Future Implications

We can then conclude that forests continue to expand in mainland Spain at a rate that is above the European average. Although the analysed period was not very long (25–30 years), changes in the conditions under which these secondary forests were being established were observable over time, i.e., in sites with greater water availability, lower elevations, less steep slopes and less forest cover. The growth trends of the forests that emerged reveal the key role of summer precipitation, temperature, slope and forest cover, and the lesser role of drought events. Finally, the results highlight the contrasting response of the diverse forest leaf-habits to these environmental factors that changed over the study period: they constrained the establishment and growth of needleleaf forests, which were most affected by warm temperatures, and favoured broadleaf secondary forests, which profited more from summer precipitation.

We consider that the general trends in secondary forest establishment and growth detected here may be extrapolated to other Mediterranean regions (e.g., Italy, Greece, Portugal) with similarities in the forest species, the biophysical conditions and their current socioecological forest situation and forestry policies [4,100,101]. The study area contains a large range of climatic, topographic and landscape conditions in the Iberian Peninsula, which in turn, is a good representative for environmental variation in the Mediterranean regions. Furthermore, autocorrelation tests confirmed the reliability of the results due to the lack of spatial structure in the models' residuals. Yet, it may be of interest to add

other factors (e.g., soil properties and forest functional composition) that may contribute to increasing the precision of the model [30,81,83,92,102] when data becomes available for global spatial and temporal analyses.

The results from this study can help to design appropriate management policies regarding forest expansion in former agricultural lands because they provide valuable information about which sites should be prioritised, e.g., where afforestation can better contribute to carbon sequestration or where it would be necessary to apply sylvicultural practices (e.g., early selective thinning) to ensure both forest productivity and long-term resilience [30,103–107]. Future research should also be aimed at studying the environmental factors that are associated with forest expansion in other climatic regions (e.g., temperate or boreal) and their effects on secondary forest establishment and growth to help with addressing the global impacts of climate change.

Supplementary Materials: The following are available online at https://www.mdpi.com/article/10.3390/land10080817/s1, Figure S1: Spearman correlation matrix for the environmental predictors potentially included to study the changes of forest establishment over the time, Figure S2: Spearman correlation matrix for the environmental predictors potentially included to study the impact on secondary forest biomass, Table S1: Classes of the Land cover maps for the analysis of global changes in the Iberian Peninsula developed by the Grumets Research Group of the Universitat Autònoma de Barcelona, Table S2: Moran's I Autocorrelation Index of secondary forest establishment, for the study sample (A) and for the residuals of the GLM with the selected explanatory variables (B), Table S3: Moran's I Autocorrelation Index of secondary forest growth (aboveground biomass; Mg ha^{-1}), for the study sample (A) for the residuals of the GLM with the selected explanatory variables (B).

Conflicts of Interest: The authors declare that they have no conflict of interest.

References

1. Song, X.P.; Hansen, M.C.; Stehman, S.V.; Potapov, P.V.; Tyukavina, A.; Vermote, E.F.; Townshend, J.R. Global land change from 1982 to 2016. *Nature* **2018**, *560*, 639–643. [CrossRef] [PubMed]
2. Foley, J.A.; DeFries, R.; Asner, G.P.; Barford, C.; Bonan, G.; Carpenter, S.R.; Chapin, F.S.; Coe, M.T.; Daily, G.C.; Gibbs, H.K.; et al. Global consequences of land use. *Science* **2005**, *309*, 570–574. [CrossRef]
3. Alkama, R.; Cescatti, A. Climate change: Biophysical climate impacts of recent changes in global forest cover. *Science* **2016**, *351*, 600–604. [CrossRef]
4. FAO. *Global Forest Resources Assessment 2020—Key Findings*; FAO: Rome, Italy, 2020. [CrossRef]
5. Meyfroidt, P.; Lambin, E.F. Global forest transition: Prospects for an end to deforestation. *Annu. Rev. Environ. Resour.* **2011**, *36*, 343–371. [CrossRef]
6. Vilà-Cabrera, A.; Espelta, J.M.; Vayreda, J.; Pino, J. "New Forests" from the Twentieth Century are a Relevant Contribution for C Storage in the Iberian Peninsula. *Ecosystems* **2017**, *20*, 130–143. [CrossRef]
7. Baśnou, C.; Álvarez, E.; Bagaria, G.; Guardiola, M.; Isern, R.; Vicente, P.; Pino, J. Spatial patterns of land use changes across a mediterranean metropolitan landscape: Implications for biodiversity management. *Environ. Manag.* **2013**, *52*, 971–980. [CrossRef]
8. FAO. Forests and agriculture: Land-use challenges and opportunities. In *State of the World's Forests*; FAO: Rome, Italy, 2016; Volume 45, pp. 811–922. [CrossRef]

9.	EEA. Environmental indicator. In *EEA Report 30/2016*; European Environment Agency: Copenhagen, Denmark, 2016; p. 60.

10.	Plieninger, T.; Gaertner, M.; Hui, C.; Huntsinger, L. Does land abandonment decrease species richness and abundance of plants and animals in Mediterranean pastures, arable lands and permanent croplands? *Environ. Evid.* **2013**, *2*, 1–7. [CrossRef]

11.	Regos, A.; Domínguez, J.; Gil-Tena, A.; Brotons, L.; Ninyerola, M.; Pons, X. Rural abandoned landscapes and bird assemblages: Winners and losers in the rewilding of a marginal mountain area (NW Spain). *Reg. Environ. Chang.* **2016**, *16*, 199–211. [CrossRef]

12.	Melero, Y.; Stefanescu, C.; Pino, J. General declines in Mediterranean butterflies over the last two decades are modulated by species traits. *Biol. Conserv.* **2016**, *201*, 336–342. [CrossRef]

13.	Breed, M.F.; Ottewell, K.M.; Gardner, M.G.; Lowe, A.J. Clarifying climate change adaptation responses for scattered trees in modified landscapes. *J. Appl. Ecol.* **2011**, *48*, 637–641. [CrossRef]

14.	Bowen, M.E.; McAlpine, C.A.; House, A.P.N.; Smith, G.C. Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna. *Biol. Conserv.* **2007**, *140*, 273–296. [CrossRef]

15.	Hooker, T.D.; Compton, J.E. Forest ecosystem carbon and nitrogen accumulation during the first century after agricultural abandonment. *Ecol. Appl.* **2003**, *13*, 299–313. [CrossRef]

16.	Pan, Y.; Birdsey, R.A.; Fang, J.; Houghton, R.; Kauppi, P.E.; Kurz, W.A.; Phillips, O.L.; Shvidenko, A.; Lewis, S.L.; Canadell, J.G.; et al. A Large and Persistent Carbon Sink in the World's Forests. *Science* **2011**, *333*, 988–993. [CrossRef] [PubMed]

17.	Fuchs, R.; Schulp, C.J.E.; Hengeveld, G.M.; Verburg, P.H.; Clevers, J.G.P.W.; Schelhaas, M.-J.; Herold, M. Assessing the influence of historic net and gross land changes on the carbon fluxes of Europe. *Glob. Chang. Biol.* **2015**, *22*, 2526–2539. [CrossRef]

18.	Espelta, J.M.; Cruz-Alonso, V.; Alfaro-Sánchez, R.; Hampe, A.; Messier, C.; Pino, J. Functional diversity enhances tree growth and reduces herbivory damage in secondary broadleaf forests, but does not influence resilience to drought. *J. Appl. Ecol.* **2020**, *57*, 2362–2372. [CrossRef]

19.	Compton, J.E.; Boone, R.D. Long-term impacts of agriculture on soil carbon and nitrogen in New England forests. *Ecology* **2000**, *81*, 2314–2330. [CrossRef]

20.	Fichtner, A.; von Oheimb, G.; Härdtle, W.; Wilken, C.; Gutknecht, J.L.M. Effects of anthropogenic disturbances on soil microbial communities in oak forests persist for more than 100 years. *Soil Biol. Biochem.* **2014**, *70*, 79–87. [CrossRef]

21.	Freschet, G.T.; Östlund, L.; Kichenin, E.; Wardle, D.A.; Freschet, G.T.; Östlund, L.; Kichenin, E.; Wardle, D.A. Aboveground and belowground legacies of native Sami land use on boreal forest in northern Sweden 100 years after abandonment. *Ecology* **2014**, *95*, 963–977. [CrossRef]

22.	Alfaro-Sánchez, R.; Jump, A.S.; Pino, J.; Díez-Nogales, O.; Espelta, J.M. Land use legacies drive higher growth, lower wood density and enhanced climatic sensitivity in recently established forests. *Agric. For. Meteorol.* **2019**, *276–277*, 107630. [CrossRef]

23.	Mausolf, K.; Leuschner, C.; Härdtle, W.; Hertel, D.; Fichtner, A.; von Oheimb, G.; Delory, B.M.; Jansen, K.; Temperton, V.M. Legacy effects of land-use modulate tree growth responses to climate extremes. *Oecologia* **2018**, *187*, 825–837. [CrossRef]

24.	Zhao, M.; Running, S.W. Drought-Induced Reduction in Global Terrestrial Net Primary Production from 2000 Through 2009. *Science* **2010**, *329*, 940–943. [CrossRef]

25.	Allen, C.D.; Macalady, A.K.; Chenchouni, H.; Bachelet, D.; McDowell, N.; Vennetier, M.; Kitzberger, T.; Rigling, A.; Breshears, D.D.; Hogg, E.H.; et al. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *For. Ecol. Manag.* **2010**, *259*, 660–684. [CrossRef]

26.	Allen, C.D.; Breshears, D.D.; McDowell, N.G. On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the Anthropocene. *Ecosphere* **2015**, *6*, 1–55. [CrossRef]

27.	Vilà-Cabrera, A.; Rodrigo, A.; Martínez-Vilalta, J.; Retana, J. Lack of regeneration and climatic vulnerability to fire of Scots pine may induce vegetation shifts at the southern edge of its distribution. *J. Biogeogr.* **2012**, *39*, 488–496. [CrossRef]

28.	Schröter, D.; Cramer, W.; Leemans, R.; Prentice, I.C.; Araújo, M.B.; Arnell, N.W.; Bondeau, A.; Bugmann, H.; Carter, T.R.; Gracia, C.A.; et al. Ecology: Ecosystem service supply and vulnerability to global change in Europe. *Science* **2005**, *310*, 1333–1337. [CrossRef]

29.	Trugman, A.T.; Anderegg, L.D.L.; Shaw, J.D.; Anderegg, W.R.L. Trait velocities reveal that mortality has driven widespread coordinated shifts in forest hydraulic trait composition. *Proc. Natl. Acad. Sci. USA* **2020**, *117*, 8532–8538. [CrossRef] [PubMed]

30.	García-Valdés, R.; Vayreda, J.; Retana, J.; Martínez-Vilalta, J. Low forest productivity associated with increasing drought-tolerant species is compensated by an increase in drought-tolerance richness. *Glob. Chang. Biol.* **2021**, *27*, 2113–2127. [CrossRef]

31.	Vayreda, J.; Martinez-Vilalta, J.; Gracia, M.; Canadell, J.G.; Retana, J. Anthropogenic-driven rapid shifts in tree distribution lead to increased dominance of broadleaf species. *Glob. Chang. Biol.* **2016**, *22*, 3984–3995. [CrossRef]

32.	Petersson, L.K.; Milberg, P.; Bergstedt, J.; Dahlgren, J.; Felton, A.M.; Götmark, F.; Salk, C.; Löf, M. Changing land use and increasing abundance of deer cause natural regeneration failure of oaks: Six decades of landscape-scale evidence. *For. Ecol. Manag.* **2019**, *444*, 299–307. [CrossRef]

33.	Lasanta, T.; Arnáez, J.; Pascual, N.; Ruiz-Flaño, P.; Errea, M.P.; Lana-Renault, N. Space-time process and drivers of land abandonment in Europe. *Catena* **2017**, *149*, 810–823. [CrossRef]

34.	Thompson, I.; Mackey, B.; McNulty, S.; Mosseler, A. *Forest Resilience, Biodiversity, and Climate Change*; Secretariat of the Convention on Biological Diversity: Montreal, QC, Canada, 2009; Volume 43, ISBN 9292251376.

35.	Tsujino, R.; Takafumi, H.; Agetsuma, N.; Yumoto, T. Variation in tree growth, mortality and recruitment among topographic positions in a warm temperate forest. *J. Veg. Sci.* **2006**, *17*, 281–290. [CrossRef]

36. Ruiz-Benito, P.; Ratcliffe, S.; Jump, A.S.; Gómez-Aparicio, L.; Madrigal-González, J.; Wirth, C.; Kändler, G.; Lehtonen, A.; Dahlgren, J.; Kattge, J.; et al. Functional diversity underlies demographic responses to environmental variation in European forests. *Glob. Ecol. Biogeogr.* **2017**, *26*, 128–141. [CrossRef]

37. Jucker, T.; Bouriaud, O.; Avacaritei, D.; Coomes, D.A. Stabilizing effects of diversity on aboveground wood production in forest ecosystems: Linking patterns and processes. *Ecol. Lett.* **2014**, *17*, 1560–1569. [CrossRef] [PubMed]

38. Vicente-Serrano, S.M.; Lopez-Moreno, J.I.; Beguería, S.; Lorenzo-Lacruz, J.; Sanchez-Lorenzo, A.; García-Ruiz, J.M.; Azorin-Molina, C.; Morán-Tejeda, E.; Revuelto, J.; Trigo, R.; et al. Evidence of increasing drought severity caused by temperature rise in southern Europe. *Environ. Res. Lett.* **2014**, *9*, 044001. [CrossRef]

39. Domingo-Marimon, C. Contributions to the Knowledge of the Multitemporal Spatial Patterns of the Iberian Peninsula Droughts from a Geographic Information Science Perspective. Ph.D. Thesis, Universitat Autònoma de Barcelona, Bellaterra, Spain, 2016; pp. 163–165. [CrossRef]

40. Vidal-Macua, J.J.; Zabala, A.; Ninyerola, M.; Pons, X. Developing spatially and thematically detailed backdated maps for land cover studies. *Int. J. Digit. Earth* **2017**, *10*, 175–206. [CrossRef]

41. González-Guerrero, O.; Pons, X. The 2017 land use/land cover map of Catalonia based on sentinel-2 images and auxiliary data. *Rev. Teledetec.* **2020**, *2020*, 81–92. [CrossRef]

42. Santoro, M.; Cartus, O. Dataset record: ESA Biomass Climate Change Initiative (Biomass_cci): Global datasets of forest above-ground biomass for the year 2017, v1. *Cent. Environ. Data Anal.* **2019**. [CrossRef]

43. ESA CCI Biomass Project. ESA CCI Biomass Product User Guide Version 1.0. 2019, pp. 1–35. Available online: http://cci.esa.int/biomass (accessed on 10 November 2020).

44. FRA. *Forest Resources Assessment: Terms and Definition FRA 2020. Working Paper 188*; FAO: Rome, Italy, 2018.

45. Palmero-Iniesta, M.; Pino, J.; Pesquer, L.; Espelta, J.M. Recent forest area increase in Europe: Expanding and regenerating forests differ in their regional patterns, drivers and productivity trends. *Eur. J. For. Res.* **2021**, *140*, 793–805. [CrossRef]

46. Gerard, F.; Petit, S.; Smith, G.; Thomson, A.; Brown, N.; Manchester, S.; Wadsworth, R.; Bugar, G.; Halada, L.; Bezák, P.; et al. Land cover change in Europe between 1950 and 2000 determined employing aerial photography. *Prog. Phys. Geogr.* **2010**, *34*, 183–205. [CrossRef]

47. Geri, F.; Rocchini, D.; Chiarucci, A. Landscape metrics and topographical determinants of large-scale forest dynamics in a Mediterranean landscape. *Landsc. Urban Plan.* **2010**, *95*, 46–53. [CrossRef]

48. Vidal-Macua, J.; Ninyerola, M.; Zabala, A.; Domingo-Marimon, C.; Pons, X. Factors affecting forest dynamics in the Iberian Peninsula from 1987 to 2012. The role of topography and drought. *For. Ecol. Manag.* **2017**, *406*, 290–306. [CrossRef]

49. Lindner, M.; Maroschek, M.; Netherer, S.; Kremer, A.; Barbati, A.; Garcia-Gonzalo, J.; Seidl, R.; Delzon, S.; Corona, P.; Kolström, M.; et al. Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. *For. Ecol. Manag.* **2010**, *259*, 698–709. [CrossRef]

50. Palmero-Iniesta, M.; Espelta, J.M.; Gordillo, J.; Pino, J. Changes in forest landscape patterns resulting from recent afforestation in Europe (1990–2012): Pre-existing forest defragmentation versus new patch proliferation. *Ann. For. Sci.* **2020**, *77*, 1–15. [CrossRef]

51. Vanderwel, M.C.; Lyutsarev, V.S.; Purves, D.W. Climate-related variation in mortality and recruitment determine regional forest-type distributions. *Glob. Ecol. Biogeogr.* **2013**, *22*, 1192–1203. [CrossRef]

52. Bonan, G. Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. *Science* **2008**, *320*, 1444. [CrossRef]

53. Li, Z.; Zhou, T.; Zhao, X.; Huang, K.; Gao, S.; Wu, H.; Luo, H. Assessments of drought impacts on vegetation in China with the optimal time scales of the climatic drought index. *Int. J. Environ. Res. Public Health* **2015**, *12*, 7615–7634. [CrossRef]

54. Ivits, E.; Horion, S.; Fensholt, R.; Cherlet, M. Drought footprint on European ecosystems between 1999 and 2010 assessed by remotely sensed vegetation phenology and productivity. *Glob. Chang. Biol.* **2014**, *20*, 581–593. [CrossRef]

55. Zargar, A.; Sadiq, R.; Naser, B.; Khan, F.I. A review of drought indices. *Environ. Rev.* **2011**, *19*, 333–349. [CrossRef]

56. Agnew, C.T. Using the SPI to Identify Drought. *Drought Netw. News* **2000**, *12*, 5–12.

57. Mishra, A.K.; Singh, V.P. A review of drought concepts. *J. Hydrol.* **2010**, *391*, 202–216. [CrossRef]

58. Reuter, H.I.; Nelson, A.; Jarvis, A. An evaluation of void-filling interpolation methods for SRTM data. *Int. J. Geogr. Inf. Sci.* **2007**, *21*, 983–1008. [CrossRef]

59. Barton, K.; Barton, M.K. Package "MuMIn.". In *Multi-Model Inference*; Version 1(6); Available online: cran.r-project.org (accessed on 8 March 2019).

60. Crawley, M.J. *The R Book: Mixed-Effects Models*; Wiley Publishing: Silwood Park, UK, 2007; pp. 627–666. ISBN 978-0-470-51024-7.

61. Gittleman, J.L.; Kot, M. Adaptation: Statistics and a null model for estimating phylogenetic effects. *Syst. Zool.* **1990**, *39*, 227–241. [CrossRef]

62. Dormann, C.F.; McPherson, J.M.; Araújo, M.B.; Bivand, R.; Bolliger, J.; Carl, G.; Davies, R.G.; Hirzel, A.; Jetz, W.; Kissling, W.D.; et al. Methods to account for spatial autocorrelation in the analysis of species distributional data: A review. *Ecography* **2007**, *30*, 609–628. [CrossRef]

63. Keenan, R.J.; Reams, G.A.; Achard, F.; de Freitas, J.V.; Grainger, A.; Lindquist, E. Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *For. Ecol. Manag.* **2015**, *352*, 9–20. [CrossRef]

64. Gold, S.; Korotkov, A.; Sasse, V. The development of European forest resources, 1950 to 2000. *For. Policy Econ.* **2006**, *8*, 183–192. [CrossRef]

65. Fuchs, R.; Herold, M.; Verburg, P.H.; Clevers, J.G.P.W.; Eberle, J. Gross changes in reconstructions of historic land cover/use for Europe between 1900 and 2010. *Glob. Chang. Biol.* **2015**, *21*, 299–313. [CrossRef] [PubMed]
66. Regos, A.; Ninyerola, M.; Moré, G.; Pons, X. Linking land cover dynamics with driving forces in mountain landscape of the Northwestern Iberian Peninsula. *Int. J. Appl. Earth Obs. Geoinf.* **2015**, *38*, 1–14. [CrossRef]
67. Terres, J.M.; Scacchiafichi, L.N.; Wania, A.; Ambar, M.; Anguiano, E.; Buckwell, A.; Coppola, A.; Gocht, A.; Källström, H.N.; Pointereau, P.; et al. Farmland abandonment in Europe: Identification of drivers and indicators, and development of a composite indicator of risk. *Land Use Policy* **2015**, *49*, 20–34. [CrossRef]
68. Kuemmerle, T.; Levers, C.; Erb, K.; Estel, S.; Jepsen, M.R.; Müller, D.; Plutzar, C.; Stürck, J.; Verkerk, P.J.; Verburg, P.H.; et al. Hotspots of land use change in Europe. *Environ. Res. Lett.* **2016**, *11*, 064020. [CrossRef]
69. Leal Filho, W.; Mandel, M.; Al-Amin, A.Q.; Feher, A.; Chiappetta Jabbour, C.J. An assessment of the causes and consequences of agricultural land abandonment in Europe. *Int. J. Sustain. Dev. World Ecol.* **2017**, *24*, 554–560. [CrossRef]
70. Fuchs, R.; Herold, M.; Verburg, P.H.; Clevers, J.G.P.W. A high-resolution and harmonized model approach for reconstructing and analysing historic land changes in Europe. *Biogeosciences* **2013**, *10*, 1543–1559. [CrossRef]
71. Vadell, E.; De Miguel, S.; Fernández Centeno, G.; Robla, E.; Lerner, M.; Pemán García, J. La forestación de tierras agrícolas: Balance de un instrumento de política forestal para el cambio del uso de la tierra. *Cuad. Soc. Española Cienc. For.* **2019**, *45*, 1–20. [CrossRef]
72. Vicente-Serrano, S.M. Differences in spatial patterns of drought on different time scales: An analysis of the Iberian Peninsula. *Water Resour. Manag.* **2006**, *20*, 37–60. [CrossRef]
73. Páscoa, P.; Gouveia, C.M.; Russo, A.; Trigo, R.M. Drought trends in the Iberian Peninsula over the last 112 years. *Adv. Meteorol.* **2017**, 4653126. [CrossRef]
74. García-Valdecasas Ojeda, M.; Gámiz-Fortis, S.R.; Romero-Jiménez, E.; Rosa-Cánovas, J.J.; Yeste, P.; Castro-Díez, Y.; Esteban-Parra, M.J. Projected changes in the Iberian Peninsula drought characteristics. *Sci. Total Environ.* **2021**, *757*, 143702. [CrossRef]
75. Gavilán, R.G.; Vilches, B.; Gutiérrez-Girón, A.; Blanquer, J.M.; Escudero, A. Sclerophyllous versus deciduous forests in the Iberian Peninsula: A standard case of Mediterranean climatic vegetation distribution. In *Geographical Changes in Vegetation and Plant Functional Types*; Springer: Berlin, Germany, 2018; pp. 101–116.
76. Terradas, J. Holm oak and holm oak forests: An introduction. In *Ecology of Mediterranean Evergreen Oak Forests*; Springer: Berlin, Germany, 1999; pp. 3–14.
77. Pasho, E.; Camarero, J.J.; de Luis, M.; Vicente-Serrano, S.M. Impacts of drought at different time scales on forest growth across a wide climatic gradient in north-eastern Spain. *Agric. For. Meteorol.* **2011**, *151*, 1800–1811. [CrossRef]
78. Vicente-Serrano, S.M. Evaluating the impact of drought using remote sensing in a Mediterranean, Semi-arid Region. *Nat. Hazards* **2007**, *40*, 173–208. [CrossRef]
79. Blanco, E.; Casado, M.A.; Costa, M.; Escribano, R.; García, M.; Génova, M.; Gómez, A.; Gómez, F.; Moreno, J.C.; Morla, C.; et al. *Los bosques ibéricos. Una Interpreación. Geobotánica*; Planeta: Barcelona, Spain, 1997; pp. 120–134.
80. Sardans, J.; Rodà, F.; Peñuelas, J. Phosphorus limitation and competitive capacities of Pinus halepensis and Quercus ilex subsp. rotundifolia on different soils. *Plant Ecol.* **2004**, *174*, 307–319. [CrossRef]
81. Gómez-Aparicio, L.; García-Valdés, R.; Ruíz-Benito, P.; Zavala, M.A. Disentangling the relative importance of climate, size and competition on tree growth in Iberian forests: Implications for forest management under global change. *Glob. Chang. Biol.* **2011**, *17*, 2400–2414. [CrossRef]
82. Coll, M.; Peñuelas, J.; Ninyerola, M.; Pons, X.; Carnicer, J. Multivariate effect gradients driving forest demographic responses in the Iberian Peninsula. *For. Ecol. Manag.* **2013**, *303*, 195–209. [CrossRef]
83. Carnicer, J.; Barbeta, A.; Sperlich, D.; Coll, M.; Penuelas, J. Contrasting trait syndromes in angiosperms and conifers are associated with different responses of tree growth to temperature on a large scale. *Front. Plant Sci.* **2013**, *4*, 1–19. [CrossRef] [PubMed]
84. Bartletta, M.K.; Klein, T.; Jansen, S.; Choat, B.; Sack, L. The correlations and sequence of plant stomatal, hydraulic, and wilting responses to drought. *Proc. Natl. Acad. Sci. USA* **2016**, *113*, 13098–13103. [CrossRef] [PubMed]
85. Klein, T. The variability of stomatal sensitivity to leaf water potential across tree species indicates a continuum between isohydric and anisohydric behaviours. *Funct. Ecol.* **2014**, *28*, 1313–1320. [CrossRef]
86. Hacke, U.G.; Sperry, J.S.; Pockman, W.T.; Davis, S.D.; McCulloh, K.A. Trends in wood density and structure are linked to prevention of xylem implosion by negative pressure. *Oecologia* **2001**, *126*, 457–461. [CrossRef] [PubMed]
87. Klein, T.; Shpringer, I.; Fikler, B.; Elbaz, G.; Cohen, S.; Yakir, D. Relationships between stomatal regulation, water-use, and water-use efficiency of two coexisting key Mediterranean tree species. *For. Ecol. Manag.* **2013**, *302*, 34–42. [CrossRef]
88. Ferrio, J.P.; Florit, A.; Vega, A.; Serrano, L.; Voltas, J. Δ13C and tree-ring width reflect different drought responses in Quercus ilex and Pinus halepensis. *Oecologia* **2003**, *137*, 512–518. [CrossRef]
89. Laamrani, A.; Valeria, O.; Bergeron, Y.; Fenton, N.; Cheng, L.Z.; Anyomi, K. Effects of topography and thickness of organic layer on productivity of black spruce boreal forests of the canadian clay belt region. *For. Ecol. Manag.* **2014**, *330*, 144–157. [CrossRef]
90. Ming, Q.; Guo, S.; Jiao, Y. High gradient effects of forest biomass energy in mountainous region—A case of meili snow mountain. *Procedia Earth Planet. Sci.* **2011**, *2*, 315–320. [CrossRef]
91. Tateno, R.; Hishi, T.; Takeda, H. Above-and belowground biomass and net primary production in a cool-temperate deciduous forest in relation to topographical changes in soil nitrogen. *For. Ecol. Manag.* **2004**, *193*, 297–306. [CrossRef]

92. Helman, D.; Osem, Y.; Yakir, D.; Lensky, I.M. Relationships between climate, topography, water use and productivity in two key Mediterranean forest types with different water-use strategies. *Agric. For. Meteorol.* **2017**, *232*, 319–330. [CrossRef]

93. Vidal-Macua, J.J.; Ninyerola, M.; Zabala, A.; Domingo-Marimon, C.; Gonzalez-Guerrero, O.; Pons, X. Environmental and socioeconomic factors of abandonment of rainfed and irrigated crops in northeast Spain. *Appl. Geogr.* **2018**, *90*, 155–174. [CrossRef]

94. Nainggolan, D.; de Vente, J.; Boix-Fayos, C.; Termansen, M.; Hubacek, K.; Reed, M.S. Afforestation, agricultural abandonment and intensification: Competing trajectories in semi-arid Mediterranean agro-ecosystems. *Agric. Ecosyst. Environ.* **2012**, *159*, 90–104. [CrossRef]

95. Poorter, L.; Lianes, E.; Moreno-de las Heras, M.; Zavala, M.A. Architecture of Iberian canopy tree species in relation to wood density, shade tolerance and climate. *Plant Ecol.* **2012**, *213*, 707–722. [CrossRef]

96. Montoya, D.; Zavala, M.A.; Rodríguez, M.A.; Purves, D.W. Animal versus wind dispersal and the robustness of tree species to deforestation. *Science* **2008**, *320*, 1502–1504. [CrossRef]

97. Paula, S.; Arianoutsou, M.; Kazanis, D.; Tavsanoglu, Ç.; Lloret, F.; Buhk, C.; Ojeda, F.; Luna, B.; Moreno, J.M.; Rodrigo, A.; et al. Fire-related traits for plant species of the Mediterranean Basin. *Ecology* **2009**, *90*, 1420. [CrossRef]

98. Fernandes, M.R.; Aguiar, F.C.; Ferreira, M.T. Assessing riparian vegetation structure and the influence of land use using landscape metrics and geostatistical tools. *Landsc. Urban Plan.* **2011**, *99*, 166–177. [CrossRef]

99. Zhang, T.; Niinemets, Ü.; Sheffield, J.; Lichstein, J.W. Shifts in tree functional composition amplify the response of forest biomass to climate. *Nature* **2018**, *556*, 99–102. [CrossRef]

100. Kankaanpää, S.; Carter, T.R. *An Overview of Forest Policies Affecting Land Use in Europe*; The Finnish Environment Institute: Helsinki, Finland, 2004; ISBN 9521117389.

101. Metzger, M.J.; Bunce, R.G.H.; Jongman, R.H.G.; Mücher, C.A.; Watkins, J.W. A climatic stratification of the environment of Europe. *Glob. Ecol. Biogeogr.* **2005**, *14*, 549–563. [CrossRef]

102. Doblas-Miranda, E.; Alonso, R.; Arnan, X.; Bermejo, V.; Brotons, L.; de las Heras, J.; Estiarte, M.; Hódar, J.A.; Llorens, P.; Lloret, F.; et al. A review of the combination among global change factors in forests, shrublands and pastures of the Mediterranean Region: Beyond drought effects. *Glob. Planet. Chang.* **2017**, *148*, 42–54. [CrossRef]

103. Duveneck, M.J.; Scheller, R.M.; White, M.A. Effects of alternative forest management on biomass and species diversity in the face of climate change in the northern Great Lakes region (USA). *Can. J. For. Res.* **2014**, *44*, 700–710. [CrossRef]

104. Vilà, M.; Vayreda, J.; Comas, L.; Ibáñez, J.J.; Mata, T.; Obón, B. Species richness and wood production: A positive association in Mediterranean forests. *Ecol. Lett.* **2007**, *10*, 241–250. [CrossRef]

105. Phillips, R.P.; Ibáñez, I.; D'Orangeville, L.; Hanson, P.J.; Ryan, M.G.; McDowell, N.G. A belowground perspective on the drought sensitivity of forests: Towards improved understanding and simulation. *For. Ecol. Manag.* **2016**, *380*, 309–320. [CrossRef]

106. Vayreda, J.; Martinez-Vilalta, J.; Gracia, M.; Retana, J. Recent climate changes interact with stand structure and management to determine changes in tree carbon stocks in Spanish forests. *Glob. Chang. Biol.* **2012**, *18*, 1028–1041. [CrossRef]

107. Cameron, A.D. Importance of early selective thinning in the development of long-term stand stability and improved log quality: A review. *Forestry* **2002**, *75*, 25–35. [CrossRef]