Land-use legacies and climate change as a double challenge to oak forest resilience: mismatches of geographical and ecological rear edges

### Short title: *Relict oak resilience at the rear edge*

### Authors[[1]](#footnote-1)

A.J. Pérez-Luque1,2,a; Gea-Izquierdo, G.3,b and Zamora, R.1,2,c

1Instituto Interuniversitario de Investigación del Sistema Tierra en Andalucía (CEAMA), Universidad de Granada. Avda. del Mediterráneo s/n, E-18006 Granada, Spain. 2Grupo de Ecología Terrestre, Departamento de Ecología, Facultad de Ciencias, Universidad de Granada, Avda. Fuentenueva s/n, E-18071 Granada, Spain. 3INIA-CIFOR. Ctra. La Coruña km 7.5. E-28040 Madrid, Spain

a [ajperez@ugr.es](mailto:ajperez@ugr.es) b [gea.guillermo@inia.es](mailto:gea.guillermo@inia.es) c [rzamora@ugr.es](mailto:rzamora@ugr.es)

### Manuscript highlights

* *Quercus pyrenaica* rear-edge forests showed high resilience at tree and stand levels
* Resilience and growth response to climate followed a water-stress gradient
* Trees and stand expressed high sensitivity to drought and land-use legacies

## Abstract

* Global change challenges ecosystems in dry locations transformed by intensive human use. Resilience to drought of relict Mediterranean *Quercus pyrenaica* Willd. populations in the southern Iberian Peninsula was analyzed in relation to historical records of land use, employing dendroecological growth of adult trees and greenness (EVI) as proxies for secondary and primary growth.
* The growth trends reflected a strong influence of land-use legacies (*e.g.* firewood removal) in the current forest structure. Trees were highly sensitive to moisture availability but both primary and secondary growth expressed high resilience to drought events over the short and the long term. Resilience and the tree growth response to climate followed a water-stress gradient. A positive growth trend since the late 1970s was particularly evident in mesic (*i.e.* colder and wetter) high-elevation stands, but absent in the most xeric (*i.e.* warmer and drier) site.
* The high values of resilience observed in our study suggest that *Quercus pyrenaica* populations in Sierra Nevada are located in a geographical but not a climatic, ecological rear edge. These resilience responses of oak forest to drought events are not spatially homogeneous throughout the mountain range, due to differences in ecological conditions and/or past-management legacies. This is particularly relevant for rear-edge populations where topographic and biophysical variability can facilitate the existence of refugia

### Keywords

extreme drought, resilience, rear edge, *Quercus pyrenaica*, tree growth, dendroecology, remote sensing

## Introduction

The response of species to changing environments (*e.g.* distributional shifts) is likely to be determined largely by population responses at range margins (Hampe and Petit 2005). Peripheral populations are usually considered more vulnerable compared with populations occurring at the center of a species’ range (*i.e.* center-periphery hypothesis; Sagarin and Gaines 2002; Pironon and others 2016). It has been often assumed that geographically marginal populations represent ecological marginal populations. This means lower performance, higher vulnerability, and thus higher risk of extinction than for populations at the core of the species’ range (Rehm and others 2015; Pironon and others 2016; Vilà-Cabrera and others 2019; Oldfather and others 2020). Nonetheless, recent reviews report that species- and population-specific responses do not always support this hypothesis (Sexton and others 2009; Abeli and others 2014). In this respect, to fully understand changes in distribution and abundance of species as a consequence of global-change, it is crucial to determine the factors that cause mismatches between the geographical and ecological rear-edge (Vilà-Cabrera and Jump 2019).

Limits of species distribution are strongly determined by climatic factors and biotic interactions (Sexton and others 2009; Gaston 2009). Climate change is expected to cause major shifts in the distribution and abundance of plant communities, and there are already signs that more intense and longer droughts are altering forest dynamics (Allen and others 2010). Drought frequency and severity have increased in recent decades, with a trend towards drier summers, particularly for Southern Europe (Vicente-Serrano and others 2014; Stagge and others 2017). In this climatic-change context, population loss and range retractions are expected in boreal, temperate, and Mediterranean species at the lowest latitudes, elevations, and drought-prone areas of a species’ distribution, *i.e.* the rear edge (Hampe and Petit 2005). In these cases, populations are likely to be more sensitive to minor climatic and microtopographic variations (Hampe and Petit 2005), and therefore the effects of droughts are expected to be particularly relevant.

In a real world, it’s important not overlook that anthropic-actions could be a driver of change as relevant as natural drivers (Sala et al. 2000), particularly for Mediterranean areas (eg. Navarro-González and others 2013; Doblas-Miranda and others 2017). In these areas, with a long history of landscape alteration, the susceptibility and response of ecosystems to natural disturbances are conditioned by legacies of historical activity (e.g. Munteanu and others 2015; Mausolf and others 2018). The past land-use legacies interact with natural disturbances and may confound the interpretation of these disturbances (Foster and others 2003). For example, recent works showed that a quarter of current forests in the Iberian Peninsula, the rear edge of several temperate and boreal tree species, are growing on former agricultural and grazing land abandoned after the 1950s (Vilà-Cabrera and others 2017). Consequently, anthropogenic habitat modification and its legacies represent a critical dimension of marginality as they may intensify, confound or delay climate‐driven population decline at rear edges (Vilà-Cabrera and Jump 2019). In this context, the main focus of our work is to identify the impacts and responses to natural (severe drought) and anthropogenic (e.g cutting) disturbances on oak forest at the southern geographical range. Thus, the incorporation of historical perspectives aid the interpretation of the ecosystem responses to natural disturbances (Foster 2003), and allow to contextualize the response of marginal rear edge populations (Vilà-Cabrera and others 2019). In this respect, we developed a rationale that integrates the ecological, geographical and anthropogenic dimensions of marginality to determine the regional and local‐scale mechanisms shaping the probability of persistence (or extinction) of rear edge populations.

The assessment of resilience to climate and human disturbances provides critical information concerning the capacity of the forests to maintain their structure and render valuable ecosystem services. Resilience is the capacity of an ecosystem to persist and maintain its state and functions in the face of disturbance (Holling 1973; Hodgson and others 2015). Lloret et al (2011) proposed an approach which decomposes resilience to drought in three components: resistance to drought, recovery after drought and resilience. Forest resilience is determined by the capacity to reduce the impact (resistance) and the capacity to recover from the impact of disturbance (recovery) (Ingrisch and Bahn 2018). This approach has been very popular to assess the forest resilience, because it allows a simple, yet highly efficient assessment of short-term responses of trees to drought, while not exempt from some criticism (Schwarz and others 2019). A combination of several approaches, such us dendroecology and remote-sensing, allows a more complete assessment of resilience to disturbances. Dendroecological estimates of growth (*i.e.* tree-ring width) are commonly used proxies to characterize tree vitality, and annual tree-ring widths can be used to study growth changes in response to drought among individual trees (Fritts 1976). Remote sensing can be used to analyze the impact of drought on ecosystems and hence on the stand (*e.g.* Zhang and others 2013). The combination of remote sensing and dendroecology has been used to assess the effects of droughts on vegetation along ecological gradients (*e.g.* Vicente-Serrano and others 2013; Coulthard and others 2017), and also to evaluate growth resilience to drought in several tree species (*e.g.* Gazol and others 2018; Peña-Gallardo and others 2018). Nonetheless, it is crucial to ascertain whether the responses at the tree level differ from those at the ecosystem level and characterize the spatial variability of this response in rear-edge populations.

In the present study, we used remote-sensing information and dendroecological methods to evaluate the impact of drought in both canopy greenness (as a proxy to primary growth) and radial tree growth (as a proxy for secondary growth) of *Quercus pyrenaica* Willd. (Pyrenean oak) in southern relict forests at the rear-edge of the species distribution and where species performance is considered to be severely threatened by climate change. We also assessed the resilience of these forests both to several extreme drought episodes and to climate change (*i.e.* warming) over the long term (in the last few decades). Additionally we conducted a review of historical documents that explain the forest history of the study systems. Our main hypothesis is that these stands will show low resilience to extreme drought from climate change along a small-scale gradient. To test this hypothesis, we: (*i*) quantified how recent extreme drought events influenced primary and secondary growth of *Quercus pyrenaica* forests at their present geographical rear edge; (*ii*) analyzed the long-term resilience of these forests to extreme drought events, using time-series of radial growth; (*iii*) and examined differences in the resilience metrics between populations located in contrasting ecological conditions (*i.e.* xeric *vs.* mesic) along environmental gradients within the rear edge in order to detect vulnerability to climate change at the small spatial scale. For the latter task, we characterized the variability in the forest response to drought within the current geographical rear edge and assessed whether the effect of aspect and environmental conditions expressed in northern and southern populations of Pyrenean oak forests differ in their resistance, resilience, and recovery to extreme drought events.

## Materials and methods

### Tree species and study site

*Quercus pyrenaica* forests extend throughout south-western France and the Iberian Peninsula, reaching their southern limit in mountain areas of northern Morocco (Franco 1990). In the Iberian Peninsula, these forests occupy siliceous soils under meso-supramediterranean and mesotemperate areas and subhumid, humid, and hyperhumid ombroclimate. Pyrenean oak is a deciduous species that requires over 650 mm of annual precipitation and some summer precipitation. As a submediterranean species, it has lower drought tolerance than evergreen Mediterranean taxa (Río and others 2007).

The forests of this species reach their southernmost European limit in Andalusian mountains such as Sierra Nevada (37°N, 3°W), a high-mountain range with elevations of up to 3482 m *a.s.l.*. The climate is Mediterranean, characterized by cold winters and hot summers, with pronounced summer drought but with marked variability according to elevation. Sierra Nevada is considered a glacial refuge for deciduous *Quercus* species (Olalde and others 2002). Eight Pyrenean oak patches (2400 ha) have been identified in this mountain range (Figure 1), from 1100 to 2000 m *a.s.l.* and often associated with major river valleys. Today, *Quercus pyrenaica* woodlands in this mountain region represent a rear edge of their habitat distribution (Hampe and Petit 2005). They are the richest forest formation in vascular plant species of Sierra Nevada, containing several endemic and endangered plant species (Lorite and others 2008). These relict forests have undergone intensive human use throughout history (Camacho-Olmedo and others 2002). Furthermore, the conservation status of this species for southern Spain is considered “Vulnerable” and it is expected to suffer from climate change, reducing its suitable habitats in the near future (Gea-Izquierdo and others 2013).

### Climatic data and drought episodes

Climate data were obtained from the European Daily High-Resolution Observational Gridded Dataset (E-OBS v16) (Haylock and others 2008). Monthly precipitation and minimum and maximum temperatures had a 0.25 x 0.25 º resolution for the 1950-2016 period. Grid cells were selected to cover each sampled site. The SPEI (Standardized Precipitation-Evapotranspiration Index) (Vicente-Serrano and others 2010) index with a temporal scale of 6 months was used to characterize the drought conditions for the period 1961-2014.

The Iberian Peninsula underwent several extreme drought episodes in the last three decades (*e.g.* 1995, 1999, 2005, 2012; Vicente-Serrano and others 2014). The 2005 and 2012 drought events have been documented as being among the worst in recent decades for the southern Iberian Peninsula (Páscoa and others 2017), appearing as extreme drought in our climatic data (Figure S1; Table S3). We focused on these two drought events because they were included in the period having remote-sensing information of high spatial resolution (MODIS started on 2000; see below). Nevertheless, for radial growth-time series, a greater number of older drought events were also analyzed to contextualize the results for 2005 and 2012 and to evaluate forest resilience to drought over a longer term (see Table S3). A drought event was identified using the SPEI 12-months scale, following a procedure similar to the one proposed by Spinoni and others (2015). We used 0.5º grid cells covering Sierra Nevada taken from the Global SPEI Database (<http://spei.csic.es/database.html>). A severe drought event starts when SPEI falls below the threshold of -1.28 (Páscoa and others 2017; Spinoni and others 2017). A drought event is considered only when SPEI values fall below that threshold for at least two consecutive months. For each drought event, we computed: the *duration* as the number of consecutive months with the SPEI lower than a certain threshold; the *severity* as the sum of the absolute SPEI values during the drought event; the *intensity* and the *Lowest SPEI* refer to the mean and lowest value of SPEI respectively during the drought event.

### Greenness data to assess ecosystem resilience

Vegetation greenness of *Quercus pyrenaica* was characterized by means of the *Enhanced Vegetation Index* (EVI), derived from MOD13Q1 product of the MODIS sensor. EVI data consists of 16-day maximum value composite images (23 per year) of the EVI value with a spatial resolution of 250 m x 250 m. MODIS EVI data were compiled for the period 2000 - 2016. We selected the pixels covering the distribution of *Quercus pyrenaica* forests in Sierra Nevada (*n* = 928 pixels). Any values affected by clouds, snow, shadows or high content aerosols, were filtered out following recommendations for mountain regions (Reyes-Díez and others 2015).

The mean Annual EVI () as a surrogate of mean annual primary production was computed for each pixel for the period 2000 - 2016. The EVI standardized anomaly () was computed pixel-by-pixel, in order to minimize bias in the evaluation of anomalies and to provide more information concerning their magnitude (Samanta and others 2012). For each pixel, an annual EVI value was calculated by averaging EVI valid values. Then, the standardized anomaly was computed as: , where is the EVI standardized anomaly for year ; the annual mean value of EVI for year ; the average of the annual EVI values for the period of reference 2000-2016 (all except year ); and the standard deviation for the reference period. Each pixel was categorized according the EVI standardized anomalies as “greening” (), “browning” () or “no-changes” ()(Samanta and others 2012).

Rather than other vegetation indices such as the NDVI, was chosen because it is highly stable under the use of any filter (Reyes-Díez and others 2015) and because it showed highly significant correlations with annual (= 0.81) and seasonal EVI values (= 0.76 and = 0.88).

### Field sampling and dendroecological methods to assess individual tree resilience

Trees were sampled during the autumn of 2016 at two locations in contrasting N-S slopes of Sierra Nevada: San Juan (SJ), a xeric site located at the northern aspect; and Cáñar (CA), a wetter site located at the southern aspect (Figure 1; Table 1). For the southern site, two elevations were sampled: CA-Low (around 1700 m) and CA-High (around 1860 m), constituting the current low-elevational limit (CA-Low) and the tree-line (CA-High), respectively, in the site sampled. Despite the proximity of these two elevations (less than a 200-m difference) the stands differ markedly in their structure and characteristics (Table 1). The three sampling sites followed a moisture gradient: SJ < CA-Low < Ca-High (Table 1). All the sites were oak monospecific and representative of the population clusters identified for the species in this mountain range (Pérez-Luque and others 2015b). At each site, between 15 and 20 trees from either the single dominant-codominant layer in CA or the open canopy in SJ were randomly sampled. Two cores of 5 mm in diameter were taken from each tree at breast height (1.3 m) using an increment borer. Diameter at breast height (DBH) and total height were measured for each tree. In addition, stand competition affecting target trees was assessed by recording distance, azimuth, DBH, species, and total height of all neighboring living trees with DBH > 7.5 cm within a circular plot with a 10-m radius. Several competition indices were calculated: the distance independent indices *density* (), and *basal area* (BA, ); and the distance dependent index size ratio proportional to distance as (see Gea-Izquierdo and Cañellas 2009 for more details). Differences between sites for height, DBH, and competition indices were analyzed using non-parametric Kruskal-Wallis rank sum tests. When significant differences were detected, multiple comparisons were run using the Dunn’s-test with Bonferroni adjustment to correct for significance.

Tree cores were air dried, glued onto wooden mounts, and sanded. Annual radial growth (ring width, RW) was determined with a measuring device coupled to a stereomicroscope, for an accuracy of 0.001 mm. Individual ring series were first visually and statistically cross-dated with TSAP software (Rinntech, Heidelberg, Germany), using the statistics Gleichläufigkeit (GLK), t-value and the crossdating index (CDI). Cross-dating validation was finally verified using COFECHA (Holmes 1983).

The growth trends were analyzed at different time scales. To study the growth response to the inter-annual variability of climate (short-term response), pre-whitened residual chronologies (RWI) were used. These were calculated from ratios between raw growth measurements and individual cubic splines with a 50% frequency cutoff at 30 years (Fritts 1976). Tree-ring width series were standardized and detrended using dplR (Bunn 2010). Mean residual site chronologies were established by computing the biweight robust mean of all prewhitened growth indices for the trees of the same site (Fritts 1976). The statistical quality of each chronology was checked via the expressed population signal (EPS). A threshold value of EPS > 0.85 was used to determine the cutoff year of the time span that could be considered reliable.

The long-term growth response was analyzed using basal area increment (hereafter BAI, ). In theory, BAI represents a more accurate indicator of growth than ring width, since it removes variation in growth attributable to increasing stem circumference after 30-40 years of juvenile growth (Biondi and Qeadan 2008). Raw ring widths and measured DBH were used to compute BAI (Piovesan and others 2008) with the following equation: where is the radius of the tree and is the year of tree-ring formation. For each individual tree, a mean BAI series was calculated. Then, mean site BAI chronologies were determined by averaging individual tree BAI time series.

### Disturbance analyses

Disturbance chronologies were built using tree-ring width to identify abrupt and sustained increases (release events from competition) or decreases (suppressions) in radial growth (Nowacki and Abrams 1997) as indirect estimates of possible disturbance events (*e.g.* logging, drought-induced neighbor mortality) in the past. Growth changes (GC) were calculated for the individual tree-ring series using a 10-year running window as either positive (PGC) or negative (NGC) growth changes: , where is the preceding 10-year median and is the subsequent 10-year median (Rubino and McCarthy 2004).

Site-disturbance chronologies were constructed by annually averaging the individual disturbance series. To separate growth peaks caused by disturbance events and expressing stand-wise disturbances from those caused by climate, we considered a threshold of 50% of GC and more than 50% of the individual trees displaying the same growth changes (*e.g.* Gea-Izquierdo and Cañellas 2014). In addition, the history of the forest and management of our sampling sites was inferred from a detailed analysis of historical land-use changes. For this, existing historical documents were exhaustively reviewed to compile information on socio-economical activities affecting the forests being studied (Table S4). We reviewed several documentary sources: historical documents and maps; detailed mining reports; official information on recent wildfires events and forest-management practices; livestock farming; traditional irrigation channels; and studies concerning the socioeconomic dynamics of forests on Sierra Nevada at different scales (see Table S4 for references).

### Assessing ecosystem and tree individual resilience to drought

To evaluate the effects of drought events on ecosystem resilience (using greenness data) and individual tree resilience (using BAI data), we used resilience indices proposed by Lloret and others (2011). The Resistance index estimated as the ratio between performance during and before the disturbance () quantifies the severity of the impact of the disturbance in the year it occurred. The Recovery index, computed as the ratio between performance after and during disturbance (), represents the ability to recover from disturbance relative to its severity. Finally, the Resilience index () is the capacity to reach pre-disturbance performance levels. The values of these indices were computed for tree growth (BAI) and greenness (EVI mean) during each drought event. The predrought and postdrought values of each target variable (*i.e.* BAI or EVI) were computed as the mean value over a period of three years before and after the drought event, respectively. A period of three years was chosen because we found similar results on comparing periods of two, three, and four years (Figure S3b), and this time period has been used in other studies (*e.g.* Gazol and others 2018). Resilience metrics for BAI data were additionally computed for the most severe drought events since 1940 (*n* = 8; Table S3) and compared with drought severity.

### Statistical analysis

The severe drought events since 1901 were identified using SPEI-12 and regional climatic data. They were characterized in terms of duration, severity, intensity. In a first step we explored temporal trends of EVI and BAI variables. Temporal trends of were examined at the pixel scale, using the Mann–Kendall nonparametric test (Figure S7a). Temporal trends for BAI were assessed using mean BAI chronologies. The impact of drought in greenness and growth was exploring using the EVIsa and the mean RWI site chronologies (Figures S7a). Additionally the relationships between climatic variables and tree-growth variables (RWI and BAI site chronologies) were assessed using bootstrapped Pearson’s correlations estimated using treeclim (Zang and Biondi 2015). The non-climatic disturbance impacts on tree-growth were evaluated using the site disturbance chronologies.

For each of the three resilience indices studied, we used robust two-way ANOVAs to test for differences between drought events (2005 and 2012) and the oak populations studied (northern and southern exposures). These tests were used because original and log-transformed data did not follow the assumptions of normality or homogeneity of variance (Wilcox 2012). Robust measures of central tendency (M-estimator based on Huber’s Psi) were used because they were close to the mean value in all cases (Wilcox 2012). When the robust ANOVA test was run, data were bootstrapped 3000 times and trimmed automatically to control the potential influence of outliers. *Post-hoc* differences were assessed pairwise using a similar bootstrap test. All the robust ANOVA and *post-hoc* tests were carried out using the WRS2 package. The level of significance was set to 0.05 and adjusted for multiple comparisons.

Resilience metrics of BAI were also computed for the most severe drought events since 1950, and the relationship of them to the severity of the drought were explored.

## Results

### Time trends in vegetation greenness

The analysis of time trends in greenness showed that 78.9% of the EVI pixels followed a positive trend for the 2000-2016 period. The lowest values of EVI standardized anomalies for the study period were recorded during the 2005 drought, and the minimum EVI values were expressed in the northern (dry) population (Figure 2a). A “browning” episode () was found during this drought event, whereas no changes in greenness in response to the 2012 drought were detected (Figure 2b).

### Analysis of radial-growth trends and disturbances

The trees of the southern population were older than those from the northern one. In addition, trees from the southern population at high elevation were taller and their growth was significantly greater than that of trees from the other two sites, despite the competition measured as plot basal area was greatest in CA-High (Table 1, Figure 3a). The growth and height of trees from the northern and the low-elevation southern population proved similar (Figures 3a and S3a). Only trees from the southern sites (*i.e.* the wetter exposure) showed significant positive growth trends since the late 1970s (Figure 3a), this trend being far more pronounced for the high elevation, the wetter, colder site (CA-High).

Drought events reduced radial growth for all sites (Figure S2a). The strongest reduction in radial growth occurred in the 1995 drought (the worst drought spell in our climatic record, Table S3) for all sites. Tree-growth reduction followed a moisture gradient. The southern sites (CA-High and CA-Low) showed less tree-growth reduction than did the northern site (SJ), especially for 2005 and 2012 (Figure S2a), with the weakest growth reductions being in trees from the wettest site (CA-High).

The response of tree growth to water availability was greater than to temperatures. Cumulative precipitation of the hydrological year and seasonal SPEI values (*i.e.* for the Hydrological year, Spring and Summer) exhibited the highest (positive) relationship with growth for all populations (Figure S6a). Nevertheless, differences appeared between northern and southern populations: the positive relationship with SPEI was highest in the more xeric northern population (r > 0.6 *vs.* r < 0.5; Figure S6a). ~~In addition, the spring maximum temperature was the most significant limitation for tree growth only for the southern populations (Figure S6b and S6c), whereas minimum and maximum temperatures of the current September positively influenced tree growth only in the northern population.~~

The northern site (SJ) showed two major release events (GC > 50% occurring in more than 50% of trees sampled): the first during the 1940s (the most evident) and the second in 1995-2000 (Figure 3b). These periods alternated with periods of suppression. By contrast, the two southern sites showed no release events except for CA-High at the beginning of the 1830s and no suppression events in the last 50 years.

### Resilience to drought events at the ecosystem and individual-tree levels

Resilienceand resistance varied in the same direction whereas recovery varied inversely to resilience and resistance. Resilience metrics of tree-growth for drought events since 1950 (*i.e.* shared period among the three chronologies excluding the juvenile years, Table S3) revealed a positive relationship between drought severity and recovery, significant for all oak populations (Figure 4a). A similar pattern was found for resilience but proved significant only for SJ. Importantly, non-significant patterns resulted when we excluded 1995, except for recovery in SJ (Figure S5). The trees showed the highest value of tree-growth resilience for 1995, the worst drought event in our study area, particularly SJ where our results suggest a major release event also after 1995 (Figure 3b).

During the last two drought events, resilience metrics for greenness and tree growth significantly differed between drought events (Table S1). The 2005 drought event reduced greenness and growth more than that of 2012 (Table S2) but the metrics of resilience generally covaried in the same direction during those two years. ~~Resilience and resistance values were significantly higher for 2012, the most severe event, than for 2005 in both variables (Table S2; Figure 4b).~~ ~~Thus, recovery values for greenness were higher for 2005 than for the 2012 drought event. Recovery showed a contrasting pattern for EVI and tree growth~~. For EVI, resilience and resistance values were significantly higher for 2012, the most severe event, than for 2005 (Table S2; Figure 4b); whereas recovery values were higher for 2005 than for the 2012 drought event. For BAI, the resilience, resistance and recovery values were higher for 2012 than for 2005 (Table S2, Figure 4c).

The resilience metrics calculated varied significantly between sites, except for resilience of tree growth (p = 0.534; Table S1), which was similar among the three sites. The two southern populations showed lower recovery values than did the northern site both for greenness and tree growth, but resistance and resilience values were significantly higher for the southern site (Table S2).

## Discussion

By using a combined approach of remote-sensing information and dendroecology, we have quantified the drought impact on the *Quercus pyrenaica* forests of Sierra Nevada and their resilience to several severe drought events in the recent decades. Our results indicated that these relict oak populations driven by historical land use are resilient to climate change at their present rear edge. However, resistance, resilience, and forest recovery to extreme drought events are strongly influenced by mountain exposure, local environmental conditions, and management legacies. This means that the geographical and the ecological rear edges do not necessarily match and, at a small spatial scale, tree performance varies markedly along the rear edge under climate change.

### Land-use legacies shape sensitivity to climate change of forests and the present rear edge

The review of historical documents revealed that forest clearings, firewood removal, charcoal making, and mining have strongly affected the forests on Sierra Nevada (Table S4), where an estimated historical loss of broadleaf *Quercus* species approaches 90% of the cover at medium and low elevations (Jiménez-Olivencia and others 2015). Together with the analysis of the disturbance chronologies, the observed notable differences in stand structure, tree size, and age suggest different forest histories and a different management origin (*i.e.* land-use legacy) between northern (coppice) and southern populations (high forest, open woodland). On the northern slopes of Sierra Nevada (*e.g.* SJ site), land uses have been historically distributed along an elevational gradient: grasslands and shrublands for cattle farming at the highest elevations; then forest stands with some croplands; and irrigated terraces with tree crops at the lowest elevations (Jiménez-Olivencia and others 2015). In addition, other activities such as mining must have altered the forest structure, *e.g.* the SJ site has many small mines and quarries that were exploited intermittently throughout history. The release growth event discerned for the 1940s concurs with a period of maximum mining activity in this area (1925 to 1957), during which timber use increased for mine tunnels and furnaces, these also requiring large amounts of firewood to melt the mineral (Table S4). This heavy exploitation of the neighboring forest resources must have affected a significant part of this oak woodland, as shown by growth of the remnant trees at the northern site (Figure S2b).

On the other hand, woodlands on the southern slopes (*e.g.* CA site) were mixed with a greater percentage of croplands along the elevational gradient where oaks grow (Jiménez-Olivencia and others 2015). Firewood, charcoal, and acorns have been intensively exploited at the southern sites, until at least the mid-20th century, when these activities sharply declined due mainly to rural abandonment and the use of gas and fossil fuels (Valbuena-Carabaña and Gil 2013). At the CA-High site, the only positive release event found for the earliest years could be related with conversion from closed forest to an open silvopastoral system, a common management type which has been applied in the past in Iberian oak woodlands (Cañellas and others 2004; Gea-Izquierdo and others 2011) and which has been documented for this site (Valbuena-Carabaña and Gil 2013).

The other release event observed for the SJ site during the period 1995-2000 was lower than during 1940, but also affected most trees (Figures 3b, S2b). No records of forest practices in this area over the last 30 years have been found (Bonet and others 2016), and no logging has been recorded during the period 1995 - 2000 (F.J. Cano-Manuel *personal communication*). Therefore this release might be related to natural drought-induced mortality after 1995, as has been reported for other Mediterranean tree species after severe drought (*e.g.* Peñuelas and others 2001; Lloret and others 2004). On the other hand, the strong positive correlations of SPEI with tree growth for this site show high sensitivity to water availability (Gea-Izquierdo and Cañellas 2014).

### Sensitivity (resistance) of relict oaks to recent drought events

Severe drought negatively affects both primary and secondary growth of *Quercus pyrenaica* forests. This was expressed by the observed reduction in greenness and tree growth in response to the 2005 and 2012 drought events as well as by the consistent radial-growth suppression for this oak species during extreme drought events (Corcuera and others 2006; Gea-Izquierdo and Cañellas 2014). Furthermore, the greatest reduction of tree growth was detected during the 1995 drought, a characteristic negative precipitation anomaly that caused severe and extensive damage to the Mediterranean vegetation across the Iberian Peninsula (Peñuelas and others 2001; Gazol and others 2018). As with many other forest species under Mediterranean climates, moisture availability is generally the most limiting factor driving radial growth of *Quercus pyrenaica* along its distribution range in the Iberian Peninsula (Gea-Izquierdo and Cañellas 2014). Thus, our results are consistent with those of previous studies highlighting the influence of precipitation on tree-ring growth in different oak species (*e.g.* Tessier and others 1994; Di Filippo and others 2010; Gea-Izquierdo and others 2011; García-González and Souto-Herrero 2017).

Greenness proved less sensitive to drought than did tree growth, particularly for drier sites. These findings agree with previous works showing tree growth to be a more sensitive metric of forest resilience than is net primary productivity (*e.g.* Babst and others 2013; Coulthard and others 2017; Gazol and others 2018; Peña-Gallardo and others 2018), suggesting that the growth reduction could be mediated by sink more than by source limitations (Körner 2013; Fatichi and others 2014). Tree-ring records complement remote-sensing data in longer time scales by reflecting tree-growth anomalies induced by climate or disturbance over decades to centuries (Babst and others 2017) and provide an accurate measure of growth responses to droughts (Bhuyan and others 2017; Gazol and others 2018).

Greenness and tree growth were more affected by drought events in drier northern populations than in wetter southern oak populations. For example, the northern site showed higher browning intensity than did the southern sites during the 2005 drought event, and the stronger correlations of tree-growth with SPEI (hydrological year and summer) in the northern site can be interpreted as higher sensitivity to drought at drier sites (Gea-Izquierdo and Cañellas 2014). It is well known that tree growth and tree responses to drought are site-dependent (*e.g.* soil features, tree competition; Babst and others 2013), particularly for rear-edge populations (Cavin and Jump 2017; Dorado-Liñán and others 2017b). Trees at CA-High registered higher BAI than those located at lower elevations (CA-Low and SJ; Figure 3a). This shows the high variability in the response to climate exhibited along a narrow gradient, which is especially noteworthy for southern sites, as these lie close to each other and overall both are considered to constitute the rear edge for the species.

### Relict oaks show high resilience to recent drought events and long-term climatic variability

Despite the severe drought events in recent decades (Table S3), we found a positive trend for vegetation greenness of *Quercus pyrenaica* for the last 16 years. This is consistent with previous findings stressing a recent short-term increase in primary productivity for these forests coinciding with a rather wet decade in the 2000s after a dry decade in the 1990s (Pérez-Luque and others 2015a). For tree growth, positive trends also appeared in the last decade, particularly for the southern high-elevation site (CA-High, Figure 3a). Similar long-term trends have been described for this species along its distribution range only at high-elevation wet and cold sites (Gea-Izquierdo and Cañellas 2014). This could be related to a non-linear positive effect of warming for the species at cold-limited high-elevation sites (Salzer and others 2009; Gea-Izquierdo and Cañellas 2014). Importantly, for rear edges threatened by climate change, negative growth trends would have been expected, as shown for some temperate and Mediterranean species (Sánchez-Salguero and others 2012; Camarero and others 2015b; Dorado-Liñán and others 2017a).

Although the 2012 drought event was more severe and intense than that of 2005 (Table S3), resilience values for greenness and tree growth were greater for 2012. This could be due to the different timing of the two droughts. The 2012 event was a winter drought (Trigo and others 2013) occurring earlier than the shorter 2005 drought. The latter matched the period of maximum growth for oak forests in late spring (Figure S4). This highlights the importance of the timing of the drought as a key factor determining tree recovery after drought (Camarero and others 2015a; Huang and others 2018). For tree growth, the highest values of resilience were found for the two most severe events (1995 and 1999; Table S3) and tree-growth resilience was positively related to drought severity (Figure 4a).

The high drought-resilience values reported here, coinciding with high values of genetic resilience for those forests on Sierra Nevada reported elsewhere (Valbuena-Carabaña and Gil 2013, 2017) appear to indicate the strong local adaptation of this oak. Our findings agree with those of studies showing that the assumed higher vulnerability of dry edges does not necessarily hold (*e.g.* Cavin and Jump 2017). Martínez-Vilalta (2018) pointed out the importance of local adaptation and plasticity, and also of local environmental factors on the vulnerability shown by rear-edge populations. Our results highlight the ample small-scale variability at the ecological boundary and need to better confine the rear-edge limit in our forest. All this, together with the characteristic high resprouting ability of the species, would suggest a long-term persistence of those populations (Bellingham and Sparrow 2000). It should be mentioned that we studied only adult individuals established decades or centuries ago, meaning that it needs to be assessed whether the trees express resilience at the species level or whether we would find vulnerability if we analyzed seedling regeneration, as in other Mediterranean species at their xeric limit (Castro and others 2004; Vilà-Cabrera and others 2011; Gea-Izquierdo and others 2015).

In summary, two main results stand out from this research. First, the high values of resilience observed in our study suggest that *Quercus pyrenaica* populations in Sierra Nevada are located in a geographical but not a climatic, ecological rear edge (*sensu* Martínez-Vilalta 2018; Vilà-Cabrera and others 2019). The current niche is a result of land-use changes, which further complicate the definition of potential rear edges. The high resilience values observed could also be related to stabilizing mechanisms promoting community resilience that can buffer the impact of extreme events, as has been described for other species (*e.g.* *Pinus sylvestris*, Herrero and Zamora 2014).

Second, these resilience responses of oak forest to drought events are not spatially homogeneous throughout the mountain range, due to differences in ecological conditions and/or past-management legacies. In fact, there was much small-scale variability in the response to climate within the rear edge that we had *a priori* considered in our study. Furthermore, we even found positive effects of climate change in certain stands, as discussed, in disagreement with our hypothesis of expecting oak vulnerability in the geographical rear edge studied. This suggests that the rear edge therefore needs to be redefined (Vilà-Cabrera and others 2019), partly because of land-use legacies and their effect on the possible mismatch between the current distribution of species (*i.e.* determining the “available” geographical rear edge) and the ecological (limiting) rear edge of species.

Overall, our results show that management history influences tree growth and resilience to climate change of tree species, highlighting the importance of land-use legacies in Mediterranean forests (Navarro-González and others 2013; Doblas-Miranda and others 2017) which, hence, will also strongly determine the current geographical distribution of marginal stands, regardless of the potential extent of ecological marginality of species.

## Concluding comments

The ecological and geographical rear edges did not appear to fully match in our study. Severe drought events provoke major reductions in primary and secondary growth of *Quercus pyrenaica* forests in the relict, rear-edge forest studied. However, we found no negative growth trends despite our expectation of vulnerability to climate change for these relict stands. Furthermore, we detected positive trends for primary growth (*i.e.* greenness) at the ecosystem scale and a steep positive trend of secondary growth at the tree level at the wettest site along the climatic gradient analyzed. The trees exhibited high resilience values in response to drought, particularly in the long-term scale. These findings are consistent with other results showing that this mountain region is still acting as a refuge for deciduous species, including *Quercus*. The differences found in tree growth and resilience to drought between sites close together show that responses to drought were site dependent and can drastically vary even in very narrow spatial gradients (*i.e.* following ecological thresholds). This is particularly relevant for rear-edge populations where topographic and biophysical variability facilitates the existence of microrefugia. The analysis of tree-growth dynamics revealed suppression and release events that were consistent with legacies left by land use in local forest dynamics, as inferred from an exhaustive review of historical documents. In this sense, our results highlight the importance of land-use legacies for highly transformed Mediterranean systems. This is relevant for tree species with a high sensitivity to climate change, such as *Quercus pyrenaica*, not only for conservation *per se* of the species, but for all ecosystem services that these singular forests offer.

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1. Author Contributions. AJPL, GGI and RZ conceived of the study, conducted field work, and collected the data. AJPL and GGI performed the lab work. AJPL analyzed data and led the writing of the paper. GGI and RZ contributed in the writing process. All authors contributed to the drafts and gave final approval for publication. [↑](#footnote-ref-1)