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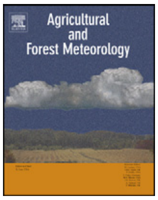

Impacts of drought at different time scales on forest growth across a wide climatic gradient in north-eastern Spain

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ABSTRACT

We analyzed the impact of drought measured on different time-scales on radial growth of eight tree species during the period 1950–2005 growing across a wide climatic gradient encompassing semiarid Mediterranean woodlands and wet mountain forests in north-eastern Spain. A drought index (standardized precipitation index, SPI) at different time scales (1–48 months) was correlated with chronologies of ring width to determine the significant time scale at which drought affected most tree growth. The findings indicated that the impact of drought on growth varied noticeably among species and sites. Two distinct patterns were clearly observed considering spatial and temporal differences in the response of species to drought. Species growing in xeric sites (*Pinus* and *Quercus* species and *Juniperus thurifera*) showed the highest responses to SPI time-scales of 9–11 months while those located in mesic sites (*Abies alba*, *Pinus sylvestris*) did respond more to SPI time scales shorter than 5 months. The SPI-growth correlations were significant, although weak, up to 30 months in xeric sites while no consistent association was observed at higher time scales. Important seasonal differences were noticed in the SPI-growth associations. Species growing in xeric areas responded to spring-summer SPI while those distributed in mesic sites responded more to summer SPI. Our findings should be useful to understand forest responses to climate change, including an increasing frequency of severe droughts, and to adapt appropriate management strategies to mitigate the impact of drought on tree growth.

1. Introduction

Water availability is one of the main climatic constraints for the tree growth in the Circum-Mediterranean forests. Thus, several studies have showed a strong correlation between precipitation and radial growth in different Mediterranean forests and tree species (Tardif et al., 2003; Macias et al., 2006; Andreu et al., 2007; Sarris et al., 2007; De Luis et al., 2009; Linares et al., 2010a; Carrer et al., 2010; Lebourgeois et al., 2010; Mérian and Lebourgeois, 2011). The impact of water deficit on growth is much higher in the most arid sites, in which the water availability largely constrains the main physiological processes of vegetation (growth, photosynthesis, carbon and nitrogen use), than in mesic sites (e.g., Vicente-Serrano et al., 2006, 2010a; Jump et al., 2006; Sarris et al., 2007; Martínez-Vilalta et al., 2008). Although the main patterns of precipitation-growth relationships are well known, the large seasonality and year-to-year variability that characterize precipitation in the Mediterranean region and the different site-dependent seasonality of tree growth in forests from this area may make very difficult to determine the response times of tree growth to the precipitation deficit. Furthermore, lags between water shortages and growth can appear as a function of different anatomical and physiological adjustments of trees to cope with drought, but also in response to drought severity and to the season in which water deficit occurs. All these mechanisms, either isolated or acting synergistically, can challenge the identification of drought impacts on tree growth.

Commonly drought indices are used with the purpose of solving the current problems of quantifying drought severity since it is very complex to determine the magnitude, duration and surface extent of droughts (Wilhite and Glantz, 1985; Redmond, 2002). Different drought indices have been developed to quantify the water deficit in an objective way, which is usually better than using the precipitation information itself (Keyantash and Dracup, 2002; Heim, 2002; Mishra and Singh, 2010). Drought indices are based on the quantification of the cumulative water shortages over a period of

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time. Some of the indices are based on soil water balance equations. The best example of this type of indices is the Palmer Drought Severity Index (PDSI, Palmer, 1965; Wells et al., 2004). Different studies have analyzed the influence of the drought conditions on tree growth using the PDSI (Orwig and Abrams, 1997; Kempes et al., 2008; Bhuta et al., 2009; Mundo et al., 2010). Nevertheless, although the PDSI can be useful to determine the severity of a drought, the index has several deficiencies (Alley, 1984; Weber and Nkemdirim, 1998), being its main shortcoming that it can only be calculated at a unique time scale (Guttman, 1998; Vicente-Serrano et al., 2010b). On the contrary, drought is a multi-scalar phenomenon, given the great variety of response times found in different hydrological, agricultural and environmental systems to the occurrence of water deficits (e.g., McKee et al., 1993; Ji and Peters, 2003; Vicente-Serrano and López-Moreno, 2005; Lorenzo-Lacruz et al., 2010; Quiring and Ganesh, 2010).

The problems involved with the use of PDSI have motivated the development of drought indices that can be calculated at different time-scales such as the standardized precipitation index (SPI) (McKee et al., 1993) and the standardized precipitation evapotranspiration index (SPEI) (Vicente-Serrano et al., 2010b). The quantification of droughts at different time scales is crucial to determine their ecological impacts, given the different physiological strategies of vegetation to cope with water deficit. Studies analyzing the drought impacts on vegetation activity using the SPI have showed contrasting responses according to the time scales at which drought affected vegetation activity and also depending on vegetation types (Vicente-Serrano, 2007) and environmental conditions for the same vegetation type (e.g., Ji and Peters, 2003; Quiring and Ganesh, 2010). At present, most studies that quantified the vegetation response to different drought time-scales have been carried out using remote sensing data for mid-term (10–30 years) datasets, which is related to the potential photosynthetic activity of the canopy or the leaf area of the forest (Vicente-Serrano, 2007). Nevertheless, presently there are a lack of studies analyzing the response of tree secondary growth to different time-scales of drought, which are quantified by means of multi-scalar drought indices. This methodological approach could improve the knowledge of the long-term responses of tree growth to water availability better than using precipitation data itself or other drought indices.

Currently, a deeper knowledge of the tree growth responses to water shortages in the Mediterranean Basin is a crucial task since General Climate Change Models predict a large reduction of precipitation and an increase of the evapotranspiration rates by the end of the twentieth-one century (Giorgi and Lionello, 2008; García-Ruiz et al., 2011). Forests are particularly sensitive to climate change because the long-life span of trees does not allow for a rapid adaptation to rapid environmental changes such as current climate warming (Andreu et al., 2007). As a matter of fact, various studies have provided evidence on the direct effect of drought on forest decline, particularly in Mediterranean forest ecosystems where water shortage is the main factor constraining growth (Sarris et al., 2007; Linares et al., 2010a; Sánchez-Salguero et al., 2010).

In this study we analyze the response of tree growth to different time scales of drought, quantified by means of the standardized precipitation index, in forests from north-eastern Spain, where a wide climatic gradient exists including diverse forest types such as Mediterranean woodlands under semiarid conditions to mountain forests under humid conditions. Furthermore, a trend towards more arid conditions (e.g., a decrease in spring precipitation) was observed during the late 20th century in the study area (González-Hidalgo et al., 2009, 2011; De Luis et al., 2010).

The objective of this study was to determine whether the use of multi-scalar drought indices is an effective way to determine the impact of the water deficit on growth, and whether this approach may detect different responses, in terms of magnitude and seasonality, in a variety of tree species and locations. The study includes eight tree species growing along the mentioned climatic gradient showing contrasting vulnerability to drought stress: four pine species (*Pinus halepensis*, *P. pinea*, *P. nigra*, *P. sylvestris*), silver fir (*Abies alba*), Spanish juniper (*Juniperus thurifera*), and two oak species (*Quercus faginea*, *Q. ilex*). These species represent species typically associated with mesic sites and humid conditions (e.g., *A. alba*), transitional locations (e.g., *P. sylvestris*) and xeric sites (e.g., *P. halepensis*).

2. Study area

The study area includes forests in Aragón, north-eastern Spain (Fig. 1). This area is subjected to Mediterranean influence with a typical summer drought and it is characterized by a strong climatic gradient ranging from semiarid conditions in the Middle Ebro Basin (mean annual temperature 15.0 °C, total annual precipitation 318 mm) to humid conditions in the Pyrenees (mean annual temperature 8.5 °C, total annual precipitation 1750 mm). In the Middle Ebro Basin the average temperatures in January and July are 6.7 °C and 26.0 °C, whereas in Pyrenean stations the means for both months are 1.5 °C and 16.8 °C, respectively (Cuadrat et al., 2007). Seasonal variability in precipitation is less pronounced in the Pyrenees than in the Ebro Basin, although summer is the predominant dry season in all study sites. In the Middle Ebro Basin there is a negative water balance (precipitation minus evapotranspiration), as a consequence of the high potential evapotranspiration (PET) that occurs in summer. Annual PET reaches 1300 mm in some sectors of this valley, which is the cause of very negative water balances. Moreover, in the Middle Ebro Basin the high temporal variability in precipitation introduces more limitations to vegetation growth, as severe droughts are frequent (Vicente-Serrano and Cuadrat, 2007), and periods of more than 80 days without precipitation are common (Vicente-Serrano and Beguería-Portugués, 2003). In the Middle Ebro Basin the lithology is characterized by millstones and gypsums (Peña et al., 2002), which contribute to aridity because there is poor retention of water by these soils (Navas and Machín, 1998). In the Pyrenees, soils are usually deep and basic and develop over limestone, sandstones and granites. The pre-Pyrenees form a mountain chain with a transitional sub-Mediterranean climate with a mean annual temperature of 11.1 °C and mean annual precipitation of about 950 mm (Cuadrat et al., 2007).

The study area contains very different forest types in terms of the dominant forest species and vegetation communities (Costa et al., 2005). Most of the studied forests are located in the mountain chains (Pyrenees and Pre-Pyrenees in the north, and the Iberian range in the south) forming pure mountain conifer (e.g., silver fir, *A. alba* Mill.) or mixed conifer-hardwood forests. In the northern study area, the Pyrenees and Pre-Pyrenees constitute transitional and mountainous areas between more humid conditions northwards or upwards and drier conditions southwards or downwards (Vigo and Ninot, 1987). In transitional areas of the Pre-Pyrenees and the Iberian range, forested landscapes comprise stands of *Pinus sylvestris* L., *Pinus pinea* L., *Pinus nigra* subsp. *salzmannii* (Dunal) Franco, *Q. faginea* Lam., and *Q. ilex* L. subsp. *ballota* (Desf.) Samp. In the semiarid Middle Ebro Basin the forests and open woodlands are dominated by *Pinus halepensis* Mill., with a few relict populations of *J. thurifera* L. The *P. halepensis* forests in the Ebro Valley commonly occur on the top and slopes of structural platforms developed on Miocene carbonate and marl sediments, whereas the valley bottoms have been traditionally used for agriculture and livestock grazing.

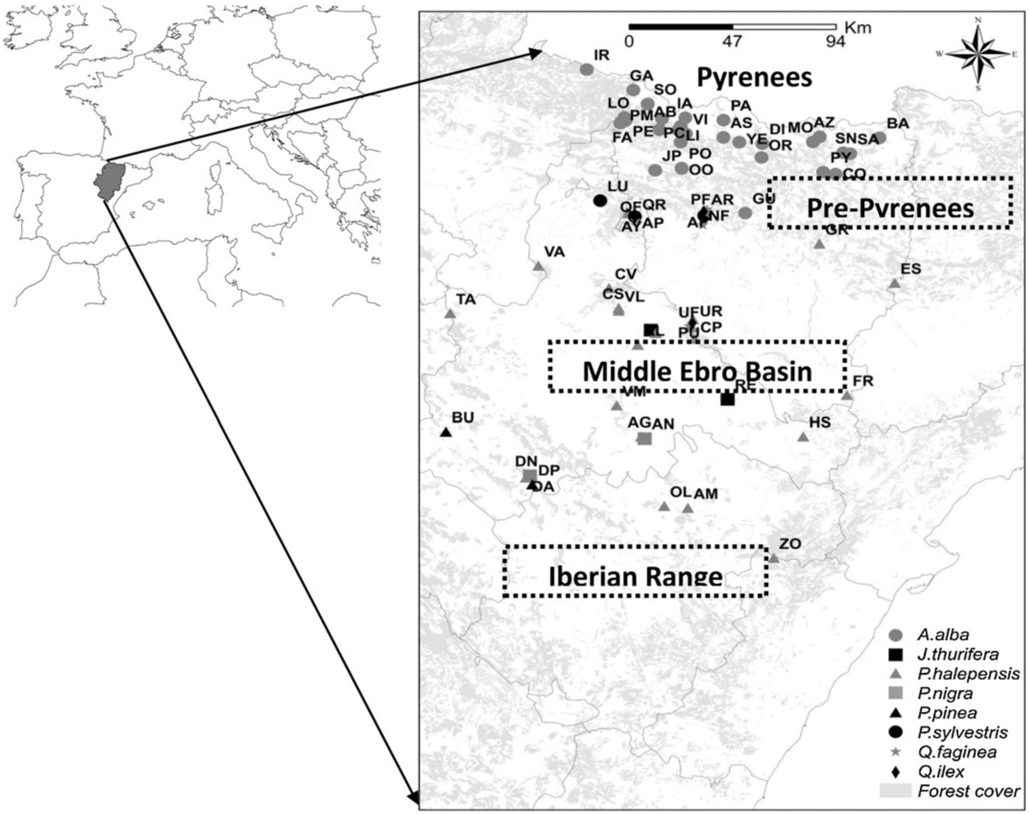

Fig. 1. Distribution of forests (gray area) in the north-eastern Spain and the location of study sites. Different symbols represent different forest species.

3. Methodology

3.1. Dendrochronological methods

Sites were selected based on the dominance of each species in the canopy over at least 1 hectare of fully forested area and the occurrence of stressing environmental conditions for tree growth such as steep slopes or shallow or rocky soils. The selected sites were considered to capture most of the climatically mediated growth variability of the studied species in north-eastern Spain. At each of the 67 sampled sites, 10–35 dominant trees were randomly selected and sampled (Table 1). At least two radial cores per tree were extracted at 1.3 m height using a Pressler increment borer. The cores were prepared following standard dendrochronological methods (Fritts, 2001). They were mounted and sanded until tree-rings were clearly visible with a binocular microscope. All samples were visually cross-dated and the ring width was measured to a precision of 0.001 mm and accuracy of ±0.0003 mm, using a LINTAB measuring device (Rinntech, Heidelberg, Germany). Cross-dating was evaluated using the COFECHA program (Holmes, 1983).

Each ring-width series was double-detrended using a negative exponential function and a spline function with a 50% frequency response of 32 years to retain high-frequency variability. Detrending and standardization involved transforming the measured values into a dimensionless index by dividing the raw values by the expected values given by the spline function. Autoregressive modeling was carried out on each series to remove temporal autocorrelation. The indexed residual series of all trees within each site were then averaged using a biweight robust mean to obtain mean site residual chronologies of ring width. We used the program ARSTAN to obtain the residual chronologies of ring width (Cook, 1985) which were used in all subsequent analyses. The quality of the chronologies was evaluated for the common period 1950–1999 using several dendrochronological statistics (Briffa and Cook, 1990): the mean width and standard deviation (SD) of the raw ring width series; the first-order autocorrelation (AC1) of raw ring width series, which measures the year-to-year persistence; the mean sensitivity (MSx) of the residual series, which quantifies the relative change in width among consecutive years; the mean correlation (R_{bar}) among individual series within each site; and the expressed population signal (EPS) of residual series, which indicates to what extent the sample size is representative of a theoretical infinite population. The common period 1950–1999 was selected because all chronologies showed EPS values above the 0.85 threshold for this interval, and this threshold is widely used in dendrochronological studies (Wigley et al., 1984). The statistical characteristics of the chronologies are shown in Table 1.

3.2. Drought index calculation

Among the existing drought indices (Heim, 2002; Mishra and Singh, 2010), only the SPI and the SPEI can be obtained at different time scales. The SPI is calculated using precipitation data exclusively. The index was developed by McKee et al. (1993) to identify the varied times of response of different hydrological systems to the precipitation deficits in a better way than other indices like the PDSI. The SPEI was developed by Vicente-Serrano et al. (2010b) to include both precipitation and temperature influence on droughts by means of the evapotranspiration processes. In this study, we used the SPI to analyze the influence of drought on tree growth since the responses of forests to long-term temperature anomalies can be very complex. An increase of evapotranspiration may have contrasting effects on tree growth depending on site conditions, the forest type and the amount of precipitation received. In drought-stressed areas an increase in the evapotranspiration rates will lead to a negative influence on tree growth, whereas in humid sites high evapotranspiration rates may enhance vegetation activity and

Table 1
Dendrochronological statistics of ring-width chronologies considering the 67 sites of the studied species for the common period 1950–1999.

Species	Site	Code	No. trees (no. radii)	MW (mm)	SD (mm)	AC1	MSx	R_{bar}	EPS
A. alba	Castiello de Jaca	CA	10 (20)	1.23	0.63	0.81	0.24	0.41	0.93
A. alba	Collubert	CO	12 (27)	2.75	1.20	0.83	0.17	0.33	0.92
A. alba	Guara	GU	10 (23)	2.92	1.18	0.70	0.20	0.55	0.96
A. alba	Orús	OR	11 (22)	1.73	0.63	0.73	0.20	0.51	0.95
A. alba	Peña Montañesa	PY	12 (21)	2.33	1.08	0.81	0.20	0.41	0.93
A. alba	Peña Oroel-high	OO	12 (24)	2.80	1.19	0.79	0.20	0.55	0.96
A. alba	Peña Oroel-low	PO	11 (23)	2.59	0.95	0.76	0.18	0.51	0.96
A. alba	San Juan de la Peña	JP	13 (28)	2.16	1.18	0.85	0.22	0.23	0.90
A. alba	Yésero	YE	12 (24)	3.44	1.43	0.77	0.20	0.43	0.94
A. alba	Irati	IR	13 (29)	1.83	0.89	0.87	0.17	0.36	0.94
A. alba	Fago	FA	11 (22)	1.34	0.73	0.71	0.29	0.54	0.97
A. alba	Paco Ezpela	PE	11 (22)	1.63	0.87	0.81	0.23	0.41	0.95
A. alba	Lopetón	LO	10 (23)	1.42	0.84	0.79	0.27	0.44	0.94
A. alba	Gamueta	GA	13 (26)	1.99	0.95	0.83	0.20	0.40	0.94
A. alba	Selva de Oza	SO	13 (25)	2.01	1.40	0.91	0.19	0.37	0.92
A. alba	Paco Mayor	PM	10 (22)	1.97	0.78	0.65	0.25	0.52	0.96
A. alba	Puente de los Corralones	PC	14 (27)	3.49	1.61	0.82	0.19	0.36	0.92
A. alba	Lierde	LI	11 (22)	2.96	1.37	0.86	0.18	0.45	0.94
A. alba	Los Abetazos	AB	12 (24)	4.46	1.95	0.97	0.16	0.33	0.90
A. alba	Izquierda del Aragón	IA	13 (25)	2.70	1.10	0.84	0.17	0.41	0.94
A. alba	Paco de Villanúa	VI	21 (42)	1.99	0.88	0.83	0.18	0.40	0.96
A. alba	Paco Asieso	AS	10 (20)	3.05	1.50	0.86	0.22	0.47	0.94
A. alba	Panticosa	PA	12 (23)	2.39	1.23	0.80	0.27	0.57	0.96
A. alba	Diazas	DI	12 (24)	2.66	1.12	0.82	0.21	0.54	0.96
A. alba	Montinier	MO	21 (30)	1.54	0.71	0.85	0.18	0.35	0.94
A. alba	Azirón	AZ	11 (22)	3.29	1.09	0.80	0.15	0.39	0.92
A. alba	Selva Negra	SN	14 (29)	2.89	1.11	0.84	0.16	0.33	0.92
A. alba	Collado de Sahún	SA	12 (29)	1.87	0.63	0.84	0.15	0.37	0.94
A. alba	Ballibierna	BA	11 (29)	2.09	0.69	0.82	0.15	0.39	0.95
P. halepensis	Agüero	AG	12 (24)	3.26	1.32	0.61	0.28	0.52	0.92
P. halepensis	Vedado de Peñaflor	PH	13 (29)	1.12	0.73	0.66	0.37	0.60	0.98
P. halepensis	Alcubierre	AL	15 (31)	1.55	1.50	0.79	0.46	0.59	0.97
P. halepensis	Alcubierre-Trincheras	AU	10 (14)	2.79	1.20	0.61	0.29	0.68	0.93
P. halepensis	Puerto de Alcubierre	PU	15 (22)	1.48	0.87	0.69	0.33	0.65	0.97
P. halepensis	Tarazona	TA	15 (30)	2.40	1.42	0.59	0.40	0.76	0.99
P. halepensis	Valareña	VA	16 (31)	1.81	1.14	0.74	0.36	0.62	0.97
P. halepensis	Castejón de Valdejasa	CV	13 (23)	3.35	1.37	0.78	0.27	0.44	0.90
P. halepensis	C. Valdejasa-Zuera	CS	12 (23)	1.14	0.73	0.60	0.43	0.68	0.98
P. halepensis	Valmadrid	VM	15 (30)	1.83	1.00	0.67	0.35	0.82	0.99
P. halepensis	Zorita	ZO	15 (29)	1.18	0.69	0.64	0.38	0.69	0.98
P. halepensis	Estopiñan del Castillo	ES	15 (27)	2.84	2.43	0.85	0.37	0.52	0.96
P. halepensis	El Grado	GR	15 (30)	2.29	1.17	0.70	0.30	0.53	0.97
P. halepensis	Oliete	OL	15 (27)	2.95	2.17	0.81	0.35	0.82	0.99
P. halepensis	Alloza	AM	15 (31)	1.45	1.00	0.72	0.39	0.73	0.99
P. halepensis	Villanueva de Gállego	VL	15 (29)	1.20	0.82	0.58	0.52	0.58	0.98
P. halepensis	Daroca	DA	14 (28)	2.06	1.17	0.50	0.44	0.82	0.99
P. halepensis	Ayerbe	AI	16 (33)	2.65	1.11	0.60	0.27	0.34	0.93
P. halepensis	Alcubierre-San Caprasio	CP	14 (27)	2.00	1.24	0.72	0.34	0.58	0.96
P. halepensis	Caspe	HS	16 (28)	0.76	0.59	0.56	0.65	0.72	0.99
P. halepensis	Fraga	FR	16 (29)	0.97	0.79	0.72	0.50	0.60	0.98
A. faginea	Agüero	QF	22 (44)	1.51	0.87	0.72	0.28	0.31	0.91
A. faginea	Arguis	AF	10 (20)	1.72	0.62	0.46	0.29	0.47	0.93
A. faginea	Nueno	NF	10 (20)	1.61	0.79	0.53	0.30	0.58	0.95
A. faginea	Pico del Águila	PF	10 (20)	1.83	0.62	0.32	0.30	0.50	0.98
A. faginea	Alcubierre	UR	33 (56)	1.61	1.00	0.43	0.44	0.70	0.99
A. ilex	Arguis	QR	10 (11)	1.34	0.56	0.37	0.34	0.32	0.91
A. ilex	Agüero	AR	10 (14)	1.16	0.60	0.32	0.44	0.46	0.92
A. ilex	Alcubierre	UR	10 (20)	0.95	0.73	0.26	0.51	0.52	0.94
J. thurifera	Retuerta de Pina	RE	35 (64)	1.09	0.71	0.61	0.42	0.56	0.90
J. thurifera	Peñaflor	PE	11 (19)	1.47	1.18	0.74	0.37	0.41	0.93
P. sylvestris	Agüero	AP	19 (37)	1.84	0.99	0.61	0.36	0.53	0.96
P. sylvestris	Luesia	LU	11 (15)	2.21	1.43	0.77	0.34	0.52	0.93
P. sylvestris	Monrepos	PS	11 (23)	1.50	0.99	0.79	0.29	0.45	0.95
P. pinea	Bubierca	BU	19 (37)	3.07	1.78	0.53	0.42	0.63	0.98
P. pinea	Daroca	DP	14 (28)	1.63	1.14	0.68	0.46	0.80	0.99
P. nigra	Agüero	AN	19 (34)	2.60	1.36	0.55	0.40	0.48	0.92
P. nigra	Daroca	DN	16 (33)	1.80	1.27	0.60	0.54	0.59	0.98

Statistics: Raw tree-ring width series: MW, mean ring width; SD, standard deviation; AC1, first order autocorrelation. Residual ring- width series: MSx, mean sensitivity; R_{bar}, mean interseries correlation; EPS, expressed population signal.

growth when soil moisture is not a constraining factor (Sabaté et al., 2002; Gaucherel et al., 2008; Vicente-Serrano et al., 2010a).

The precipitation data in the region has been obtained from a homogeneous and spatially dense dataset of daily observatories (Vicente-Serrano et al., 2010c). Nevertheless, since the available observatories are commonly located near populated areas, few stations are available in mountainous location where we sampled most of the forests. For this reason, the available punctual

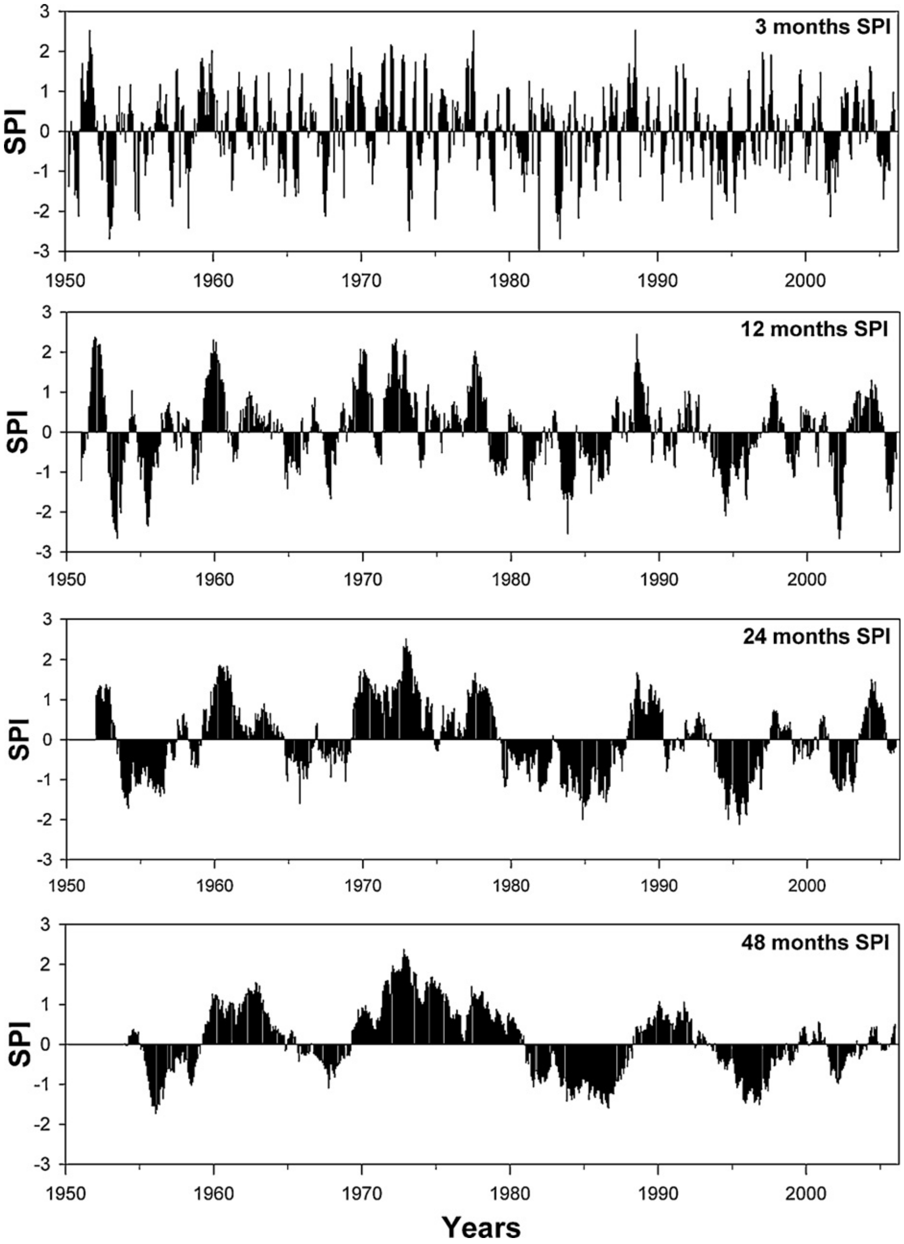

Fig. 2. Evolution of the drought index SPI at different monthly time scales in one of the forests of the study area (site AL, *P. halepensis* forest).

precipitation data was interpolated to have a regular grid with information in each one of the forests sampled. Monthly precipitation data between 1950 and 2006 was interpolated at a spatial resolution of 1000 m. To take into account the effect of the elevation on precipitation, and to have more reliable estimations for each forest, the interpolation was done using a Digital Terrain Model and a Geographic Information System (GIS)-assisted regression-based approach (Ninyerola et al., 2000, 2007; Vicente-Serrano et al., 2003, 2007). The precipitation in each 1000 m grid point was estimated, for each month between 1950 and 2006, by means of a stepwise-regression model, in which the independent variables were the elevation, the latitude and the longitude of each site. The residuals, i.e. the differences between the observations and the modeled precipitation, were also included in the estimations by means of a local interpolation procedure (splines with tension – Mitasova and Mitas, 1993) to include the local precipitation features recorded each month, which were not well represented by the regression models.

The validation of the grid layers was done for each monthly layer by a jackknifing method, based on withholding, in turn, one station out of the network, estimating regression coefficients from the remaining observatories and calculating the difference between the predicted and observed value for each withheld observatory (Phillips et al., 1992). This method has frequently been used in climatology (e.g. Daly et al., 1994; Holdaway, 1996; Hofstra et al., 2008). The average Root Mean Square Error for the different months and years was 15.2 mm, being lower in summer (6.5 mm) than in winter (22.3 mm). The D agreement index (Willmott, 1982) showed an average of 0.94 for the different monthly layers, with a range between 0.82 and 0.99, which indicates a high reliability between the observed and the modeled precipitation data.

We obtained an average monthly precipitation series for each one of the 67 sites using the gridded monthly precipitation data. From each one of the series, we obtained the SPI at time scales from 1 to 48 months for having a range of time scales to be compared with radial growth. The SPI was calculated by adjusting the

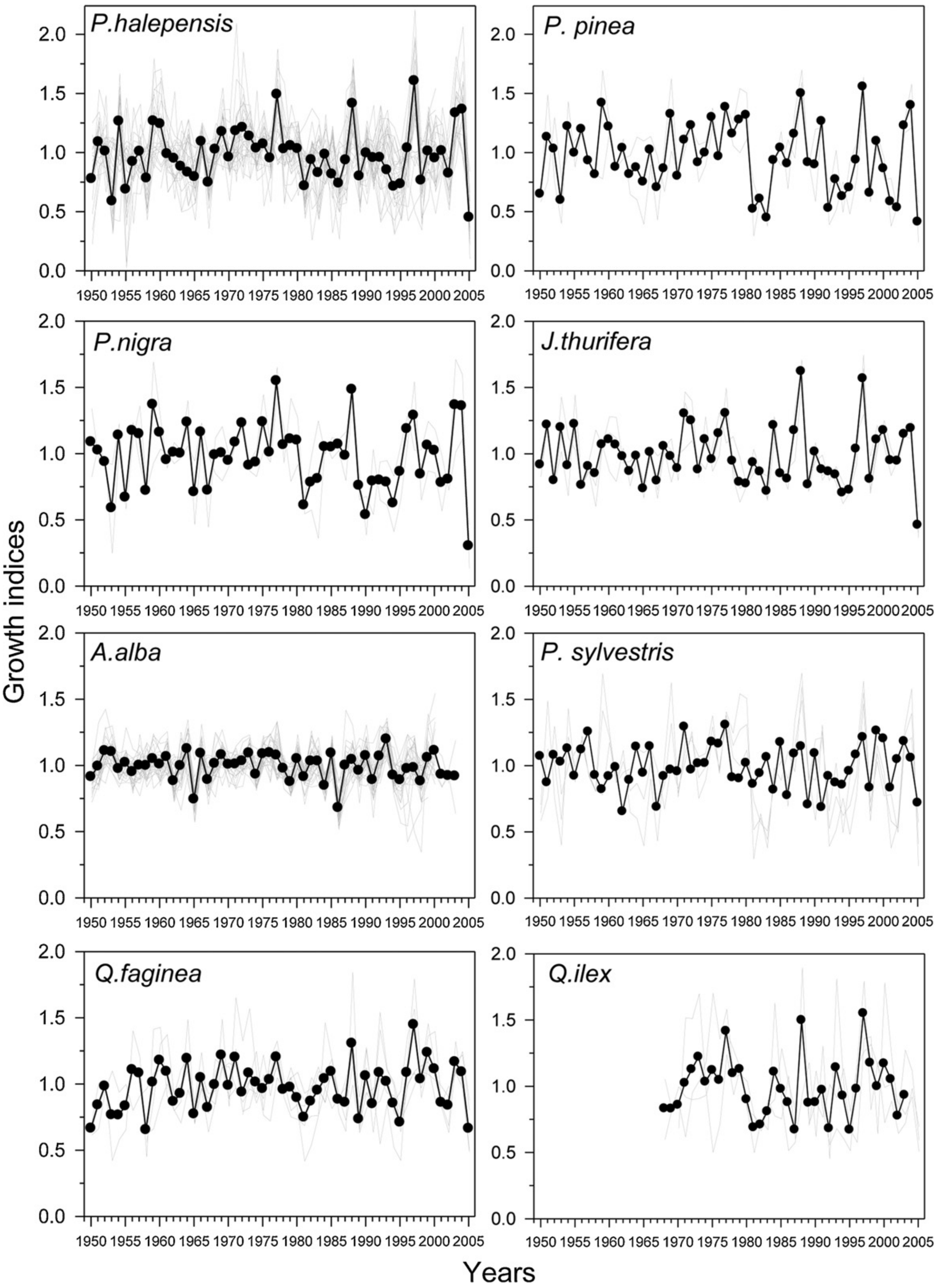

Fig. 3. Residual chronologies of ring width (gray lines) for all studied species network in north-eastern Spain, and the overall mean for each variable (black lines).

precipitation series to a given probability distribution. Initially, the Gamma distribution was used to calculate the SPI (McKee et al., 1993), but the Pearson III distribution was considered more robust due to its three parameters (Vicente-Serrano, 2006). The complete formulation of the SPI following the Pearson III distribution and the L-moments method for calculating parameters is described in Vicente-Serrano (2006) and López-Moreno and Vicente-Serrano (2008).

The Fig. 2 shows a representative example of the evolution of the SPI at the time scales of 3, 12, 24 and 48 months in one of the forest sites of the study area (site AL, *P. halepensis* forest). Droughts show high contrasted frequency as a function of the time scale. On

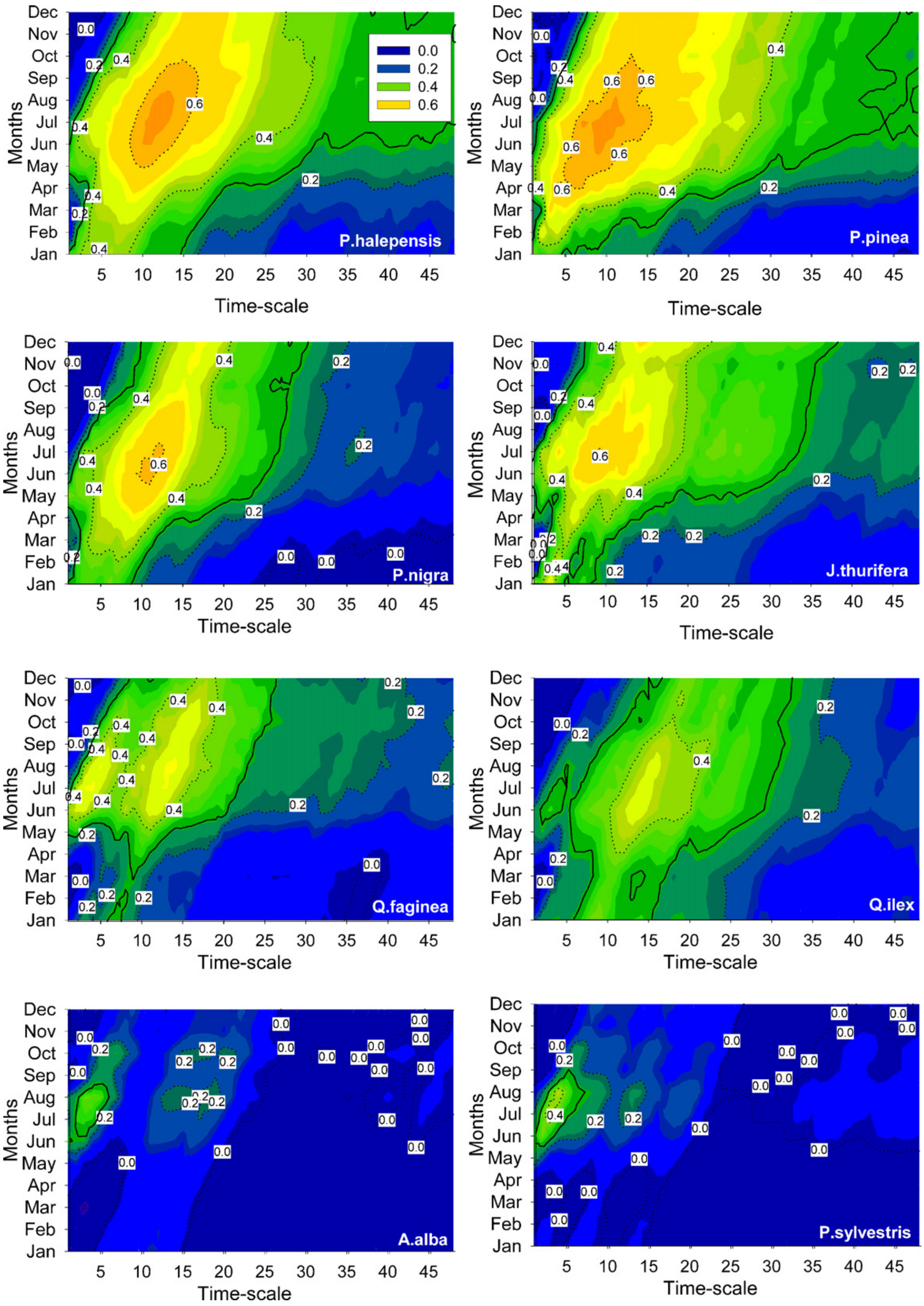

Fig. 4. Mean correlation coefficients between ring-width chronologies and monthly SPI series at different time scales and for the six species analyzed in this study. Bold lines frame significant correlations ($p < 0.05$).

the shortest time scales (e.g., 3 months), the dry and humid periods are short and occur at high frequencies. At long time scales (e.g., 24 and 48 months), droughts lasted longer, but were less frequent, with few dry or humid periods recorded.

3.3. *Statistical analyses*

To determine the influence of drought severity on tree growth, but also the drought time-scales that are affecting radial growth

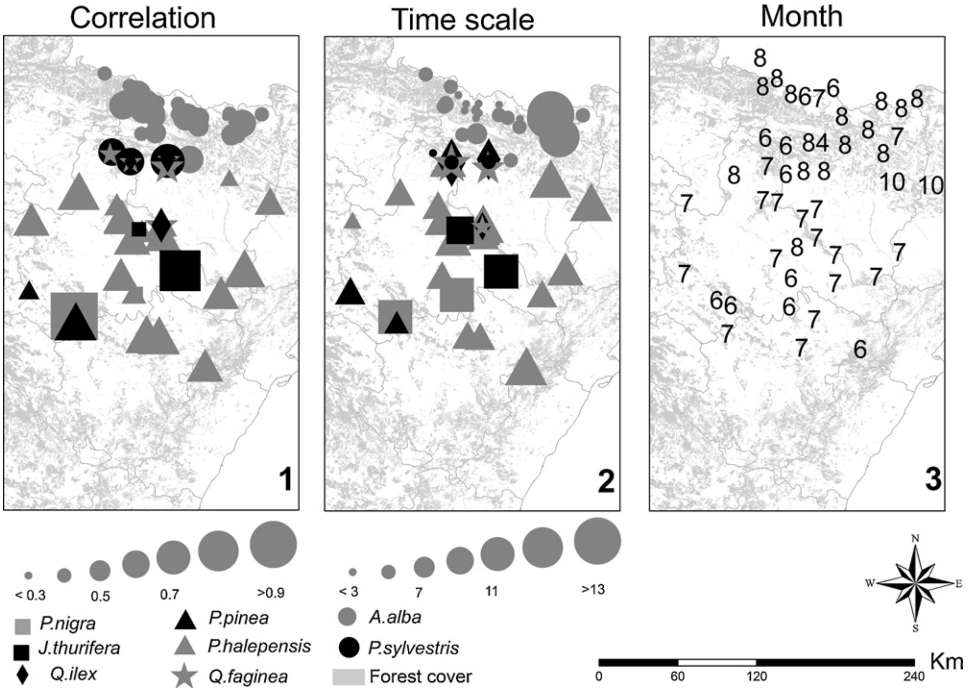

Fig. 5. Geographic variability of the growth-drought associations showing (1) maximum Pearson correlations between ring-width chronologies for all sites and the monthly SPI series (the symbols are proportional to the absolute correlation value), (2) time scale (in months) at which the maximum correlation was achieved; and (3) month of the year at which the highest correlation was reached. Symbols are as in Fig. 1.

of species and sites considered in the study, we carried out correlation analyses using the Pearson coefficient. Prior to calculating the correlations, the trend in each of the SPI time series was removed by assuming a linear evolution in each monthly series at the different time scales. Therefore, correlation analyses were performed between ring-width residual chronologies representing tree growth and detrended monthly SPI series representing drought severity for the period 1950–1999. To summarize the correlation analyses and to compare the drought impact among species, we also obtained an average correlation for all sites corresponding to a particular species. Moreover, the variability within species in terms of drought impact on growth was illustrated by selecting representative sites. The threshold for significant correlations was set at $P \leq 0.05$.

4. Results

4.1. Growth patterns

The mean ring width ranged from 1.15 mm (*Q. ilex*) to 2.39 mm (*A. alba*) across the studied dendrochronological network (Table 1). The AC1 values were highest in *A. alba* (0.81) and lowest in *Q. ilex* (0.31), while the MSx values were generally high for *P. nigra* (0.46), *P. pinea* (0.44), *Q. ilex* (0.43), *J. thurifera* (0.39) and *P. halepensis* (0.38), and low for *P. sylvestris* (0.33), *Q. faginea* (0.32) and *A. alba* (0.20). The highest R_{bar} and EPS values were observed for *P. halepensis* and the lowest were found for *A. alba*.

We observed similar high-frequency growth patterns for species from xeric sites (*P. halepensis*, *P. pinea*, *P.nigra* and *J. thurifera*) with growth reductions in 1953, 1961, 1981, 1989 and 2005 (Fig. 3). On the other hand, the periods characterized by substantial increase in radial growth included 1959, 1960, 1977, 1988 and 1997. Some of these sharp growth decreases and increases corresponded to severe droughts (e.g., 2005) and very wet years (e.g., 1997), respectively. *A. alba* showed a more stable growth pattern with growth reductions in 1965 and 1986, 1981, 1993 and 2001, while *P. sylvestris*, *Q. faginea* and *Q. ilex* showed inter annual variability in radial growth.

4.2. Species- and site-dependent associations between growth and drought

We found important differences in the responses of tree growth to the different time scales of the SPI (Fig. 4). In *P. halepensis* forests, significant correlations were found for time scales from 1 to 48 months. Nevertheless, for short time scales (1 to 3 months) significant correlations only appeared between May and July. In this specie the highest correlations ($r = 0.60$–0.70) were observed at time scales between 12 and 16 months during June, July and August. High correlations were also found between the SPI and the growth chronologies of *P. pinea* (maximum $r = 0.70$), *P. nigra* (maximum $r = 0.62$) and *J. thurifera* (maximum $r = 0.61$) considering spring (May), summer (June, July, August) and early-fall (September) months, particularly at time scales from 9 to 15 months. The remaining conifers (*A. alba*, *P. sylvestris*) did not show strong associations between growth and SPI and presented low and significant correlation coefficients ($r = 0.30$–0.40) during summer months for time scales between 2 and 5 months. A higher growth response to drought was observed for *Q. faginea* and *Q. ilex* since they showed significant correlations at time scales up to 25 months, mostly during summer and autumn months. *Q. ilex* showed lower growth-SPI correlations than *Q. faginea*, with association being significant only at time scales between 8 and 25 months. Independently of the magnitude of the association and the SPI time scale analyzed, the highest correlation coefficients between growth and the drought index were obtained in late-spring (May) and early-summer (June) months, the period when growth rates are usually maximum, and when water deficit is noticeable in most study sites.

4.3. Geographically structured growth-drought relationships

We detected a high site-to-site variability in the growth response to drought within the same species across the study region (Fig. 5). We observed a clear south-north gradient in the maximum correlations between growth and SPI. This gradient corresponded to maximum correlation values for sites located in the Middle Ebro Basin in the case of Mediterranean conifers (*P. nigra*, $r = 0.84$; *P. halepensis*, $r = 0.83$; *P. pinea*, $r = 0.81$; *J. thurifera*, $r = 0.75$) and oaks (*Q. faginea*, $r = 0.74$; *Q. ilex*, $r = 0.67$), intermediate values in sub-Mediterranean *P. sylvestris* forests from the Pre-Pyrenees ($r = 0.66$) and minimum values in Pyrenean mesic *A. alba* forests ($r = 0.64$).

Most species reached the maximum growth-drought correlations at time scales varying from 9 to 11 months, excepting *A. alba* and *P. sylvestris* which showed the maximum association at time scales lower than 5 months in most of their forests. Nevertheless, we also noted large differences of the maximum correlation time scales within species without any clear spatial pattern. Finally, irrespective of the tree species, the study site, the magnitude of the SPI-tree growth correlations and the characteristic time scale, the maximum correlations were commonly obtained in summer months (June to August), which indicates the relevance of water availability during the late growing season for tree growth.

5. Discussion

This study evaluated the impact of droughts on eight tree species forming forests distributed along a wide climatic gradient of north-eastern Spain by means of dendrochronological methods and using the standardized precipitation index. To the best of our knowledge, this is the first investigation exploring the relationship between tree growth and drought using multi-scalar drought indices. Particularly, the research has focused on the impact of different time scales characterizing drought intensity on radial growth, given that droughts may act on growth at different characteristic time scales. For instance, at short time scales dry and moist periods alter with a high frequency, but at long time scales droughts are less frequent but longer in duration (Hayes et al., 1999; Vicente-Serrano, 2006).

Our analyses revealed two distinct patterns in terms of the growth sensitivity to drought, depending on the time scale of drought and the studied species. Species growing in the Middle Ebro Basin under Mediterranean semiarid conditions (all *Pinus* species excepting *P. sylvestris*, *Quercus* species, *J. thurifera*) showed stronger growth responses to drought compared to those growing in mountainous areas from the Pre-Pyrenees and Pyrenees (*A. alba*, *P. sylvestris*) characterized by a humid and cold climate. It has been found previously that in xeric Mediterranean areas tree growth is mainly limited by low precipitation, while in mesic Mediterranean areas the main factors constraining growth are low temperatures (Richter et al., 1991; De Luis et al., 2007; Vicente-Serrano, 2007; Camarero et al., 2010). The high level of dependence on water availability of north-eastern Spanish forests has been also reported before mainly in pine and oaks species (Corcuera et al., 2004a,b; Andreu et al., 2007; Montserrat-Martí et al., 2009; Gutiérrez et al., 2011). Overall, these studies reported marked spatial variations in the response of forests to drought as a function of climate conditions, confirming that forests located in the driest sites are the most sensitive to drought occurrence.

Tree species growing in the driest sites of our study area, i.e. the Middle Ebro Basin, showed robust relationships ($r > 0.60$) with the SPI drought series at time scales between 9 and 11 months, which is a remarkable finding since it indicates that cumulative precipitation conditions during one year impact tree growth as illustrated by Sarris et al. (2007). The response of growth to drought time scales longer than 11 months decreased gradually but correlations were significant up to 30 months ($r > 0.30$). Beyond this threshold, the growth responses to drought was very low, indicating that precipitation recorded for periods longer that 30 months is not significantly affecting radial growth. Other studies that analyzed the relationships between precipitation and growth in semiarid sites showed that the correlations between two variables increased when considering the cumulative precipitation over a period of consecutive months (De Luis et al., 2009; Linares et al., 2010b). Thus, forest growth variability in similar drought-prone area is determined by the precipitation recorded during the year of tree-ring formation but also by the precipitation that fell in the previous year (Sarris et al., 2007). The use of a multi-scalar drought indicator allowed confirming this question in the analyzed forests. In the semiarid Middle Ebro Basin, the previous-winter soil water reserves are crucial for supporting tree growth during spring (Pasho et al., 2011). It has been found that spatio-temporal variations in soil moisture and related rainfall pattern determine the growth response to climate in most Iberian forests (Andreu et al., 2007). Soil types (limestone, gypsum) in the Ebro Basin valley may additionally intensify the effects of drought conditions on tree growth because they have low water holding capacity (Guerrero et al., 1999). All the studied species found in the Middle Ebro Basin showed the strongest growth response to drought during spring and summer months, which indicates that forest growth in the area is very dependent on spring and summer cumulative water deficit. First, most of these species show their maximum radial-growth rates between May and June (Camarero et al., 2010). Second, water deficit starts to affect markedly vegetation activity and plausibly growth in the study area as early as June (Vicente-Serrano, 2007). In addition, in the driest sites we found that the growth-drought correlation decreased at time scales lower than 9 months, showing that these forests may show certain adaptive capacity in response to short droughts.

The response of tree growth to drought greatly varied among species, being the maximum growth-drought correlation very high for *P. halepensis, P. pinea*, *P. nigra* and *J. thurifera* ($r = 0.60$–0.80), moderate for *Q. ilex* and *Q. faginea* ($r = 0.50$–0.60), and low for *P. sylvestris* and *A. alba* ($r = 0.40$–0.50). The variability of species responses to drought may indicate very different strategies and functional threshold in coping with droughts. The lower tree growth–drought correlations found in drought-tolerant oaks (e.g., *Q. ilex*) as compared with drought-avoiding pines (e.g., *P. halepensis*) could be related to the more efficient conductive elements (vessels in oaks *vs.* tracheids in conifers), a more conservative water and deeper root systems of the former as compared with the later species, which might mitigate the negative effects of short-term water shortages on tree growth (Hacke and Sperry, 2001; Willson et al., 2008). However, our results suggests that *Q. ilex* have showed a higher growth plasticity in response to drought, i.e. low growth-SPI correlations, in comparison to *Q. faginea*, suggesting a greater resistance to water constrains of the former as compared with the later species, which is in agreement with the low phenological activity of *Q. faginea* in summer (Corcuera et al., 2004a,b; Montserrat-Martí et al., 2009). *J. thurifera* also appeared to be affected moderately by drought despite this species is considered a drought-resistant species among the Iberian conifers and its radial-growth dynamics are very plastic in the response to drought and to episodic rains (Camarero et al., 2010).

Considering the *Pinus* species from the most arid study sites (*P. halepensis, P. pinea*, *P. nigra*), we detected a stronger response to cumulative droughts over a 11-months period and during spring and summer months in comparison to other co-existing species in the area. This indicates that tree growth in these pine species is sensitive to mid-term water deficits which agrees with the findings of Linares et al. (2010b), who found that *P. halepensis* growth in south-eastern Spain can be limited by drought during the summer prior to growth. Although pine species as *P. halepensis* are considered as drought-avoiding species, they may show functional growth thresholds in response to lasting and severe drought leading to growth decline and death (Novak et al., 2011). For instance, *P. halepensis* is adapted to the scarcity of soil water in the short-term due to summer drought by ceasing secondary growth and recovering it rapidly when water becomes available (Borghetti et al., 1998; Nicault et al., 2001; Rathgeber et al., 2005; De Luis et al., 2007, 2011). However, its growth may be much vulnerable to mid- and long-term droughts as our results support. Moreover, considering the fact that some of the studied sites represent the southernmost populations of some of the studied species in Europe (e.g., *A. alba*) and these stands are growing near the species' climatic tolerance, they may be affected severely by strong and lasting droughts leading to forest decline (Macias et al., 2006; Camarero et al., 2011).

We found no impact of long-term droughts on radial growth of mountain conifers from mesic sites (*A. alba*, *P. sylvestris*) which responded to short-term droughts of a duration lower than five months during summer. This association may be explained by the low water-use efficiency of *A. alba* which is a species whose photosynthetic rates are very sensitive to atmospheric drought (Guehl et al., 1991) and its growth rates respond to short-term cumulative water deficit in late summer (Camarero et al., 2011). In the case of *P. sylvestris* previous studies have clearly indicated that summer drought contrains growth and xylogenesis in *P. sylvestris* (Camarero et al., 2010; Gruber et al., 2010) and severe water deficit may even lead to drought-induced mortality (Martínez-Vilalta and Piñol, 2002; Sánchez-Salguero et al., 2010).

The response to drought was site-dependent and this variability among sites was greater in *P. nigra* and *Q. faginea* forests as compared with the other species. However, the number of sites sampled to capture the variability of both species was low in comparison to other well-replicated species such as *P. halepensis* and *A. alba*. In general, sites located in the driest areas of the Middle Ebro Basin showed a higher response to drought compared to those located in mesic mountainous areas where water availability is high. For example, growth in *P. halepensis* showed higher growth-drought correlations in the driest sites than in northern less xeric sites receiving more precipitation. Given the large intra-specific differences found in the study area, further studies are necessary to assess the relative roles of local conditions (topography, soil type, management history, etc.) on the growth responses to drought at different time scales particularly in those species with more sampled forests.

6. Conclusions

We have provided compelling evidence that multi-scalar drought indices are particularly useful for monitoring the impact of climate variability on forest growth because the response of tree growth to droughts is complex. The time scales over which precipitation deficits accumulate affecting noticeably forest growth vary among species and among sites within the same species. For this reason, drought indices must be associated with a specific time scale and assessed taking into account local conditions to be useful for monitoring impacts on forest growth as has been done with remote-sensing assessments of vegetation activity (Ji and Peters, 2003; Vicente-Serrano, 2007; Quiring and Ganesh, 2010).

In the current context of climate warming, several climate models have indicated that drought frequency and intensity are expected to increase in the Western Mediterranean Basin (Giorgi and Lionello, 2008). Increasing aridity is expected to cause growth decline and enhance mortality particularly in drought-sensitive species (Linares et al., 2010b; Gruber et al., 2010; Koepke et al., 2010). The approach used in this study for examining growth responses to drought at different time scales and considering multiple tree species and sites across a wide climatic gradient in north-eastern Spain may represent a first step in understanding and forecasting forest responses to future climate change.

7. Summary

The use of multi-scalar drought indices such as the standardized precipitation index is useful in determining and quantifying the drought impacts on tree growth. The eight tree species considered in this study across a wide climatic gradient in north-eastern Spain showed a high inter- and intra-specific (site-dependent) variability in terms of growth responses to drought, suggesting different mechanisms in coping with drought. Growth of tree species from in xeric sites responded strongly to mid- to long-term droughts lasting from 9 to 12 months, whereas species from mesic sites showed a lower short-term (1–5 months) response. In most cases the maximum association between growth and drought was detected from June up to August. The variability among species and sites to drought acting at different time scale could be attributed to species characteristics and local conditions of the sampled sites. If future climate warming causes more frequent and severe droughts in the future, many forests in the study area will be adversely and selectively affected. Our finding suggest the need for further studies of drought impacts on forests under a wide range of climatic conditions to provide a current spatial analogue on the influence of forecasted severe drought on tree growth and function in the future.

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