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The role of climate change in the widespread mortality of holm oak in open woodlands of Southwestern Spain

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ABSTRACT

Forest decline and increasing tree mortality are of global concern and the identification of the causes is necessary to develop preventive measures. Global warming is an emerging factor responsible for the increasing tree mortality in drought-prone ecosystems. In the southwestern Iberian Peninsula, Mediterranean holm oak open woodlands currently undergo large-scale population-level tree die-off. In this region, temperature and aridity have increased during recent decades, but the possible role of climate change in the current oak mortality has not been investigated.

To assess the role of climate change in oak die-off in managed open woodlands in southwestern Spain, we analyzed climate change-related signals in century-long tree ring chronologies of dead holm oaks. We examined the high/low-frequency variability in growth and the relationship between growth and climate.

Similar to other Mediterranean forests, growth was favored by precipitation from autumn of the year prior to ring formation to spring of the year of ring formation, whereas high temperatures during spring limited growth. Since the 1970s, the intensity of the high-frequency response to water availability increased simultaneously with temperature and aridity. The growth trends matched those of climatic changes. Growth suppressions occurred during droughts in the 1970s, 1980s and 1990s. Widespread stand-level, age-independent mortality occurred since 2005 and affected trees that cannot be considered old for the species standards.

The close relationship between growth and climate indicate that climate change strongly controlled the growth patterns. This suggests that harsher climatic conditions, especially increased aridity, affected the tree performance and could have played a significant role in the mortality process. Climate change may have exacerbated or predisposed trees to the impact of other factors (e.g. intense management and pathogens). These observations could suggest a similar future increase in oak mortality which may occur in more northern oak open woodlands if aridity further increases.

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1. Introduction

Increases in tree mortality have been recently observed in forest ecosystems worldwide (Allen et al., 2010; Choat et al., 2012). However, it is necessary to further understand the actual causes of tree mortality to assess the vulnerability of forest ecosystems and to evaluate potential preventive measures (McDowell et al., 2008; Allen et al., 2010). In the Western Mediterranean, different species show signs of decline, e.g. sustained growth reduction and defoliation (see Fig. 6 and Table 4 in Allen et al., 2010; Carnicer et al., 2011), which may indicate an increasing risk of tree mortality (Bigler and Bugmann, 2004; Gea-Izquierdo et al., 2014). In the southwest of the Iberian Peninsula, there is an alarming increase of oak mortality in managed oak open-woodlands over large areas (Carrasco, 2009). These ecosystems, dominated by evergreen holm oaks (*Quercus ilex* L.) and cork oaks (*Quercus suber* L.), are among the most representative Iberian Mediterranean landscapes and have considerable socio-economic value (Carevic et al., 2010; Alejano et al., 2011). In recent decades, certain oak stands have undergone

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a decline characterized by nonspecific symptoms, including wilting of leaves, twigs, and branches, bark necrosis, and production of epicormic shoots (Brasier, 1996; Navarro, 2011). In Southwestern Spain, this decline has extended to the regional scale in recent years and oak die-off is currently widespread. This phenomenon is a major problem for forest owners and threatens sustainability of these woodlands (Carrasco, 2009). Pathogenic fungi are a factor of oak decline (Sánchez et al., 2002). Additionally, although there is a lack of experimental evidence of the effect of management on oak mortality, management is very intense in these ecosystems (thinning, livestock management, pruning, soil tillage) and may also be involved in this decline process.

The increase in drought an heat-induced tree die-off at the global scale indicates that climate change is an emerging factor of tree mortality processes (Allen et al., 2010; Choat et al., 2012; Anderegg et al., 2013). Increases in drought can predispose trees to mortality, or directly cause tree death, through different interrelated physiological mechanisms, including carbon starvation and hydraulic failure (McDowell et al., 2008, 2011). In the Iberian Peninsula, climate has become drier and warmer in recent decades (Sumner et al., 2003; Rodrigo and Trigo, 2007; Kovats et al., 2014). In Southern Spain, evergreen oaks grow under Mediterranean-type climates where meteorological drought (i.e. period in which $P < 2T$, where “P” is mm of monthly precipitation and “T” is mean temperature in Celsius degrees) lasts for up to 5 months (Vázquez-Piqué, 2011) and is the most important limiting factor for vegetation. Moreover, narrow soils in many stands can amplify the impact of extreme climatic events like severe droughts (David et al., 2007; de Sampaio e Paiva Camilo-Alves et al., 2013). Atmospheric warming and climatic instability can increase pathogen activity and aggravate oak diseases (Brasier and Scott, 1994; Sánchez et al., 2002; Corcobado et al., 2013). Research indicates that climate change is a factor involved in forest mortality in some Spanish forests at higher latitudes (Martínez-Vilalta and Piñol, 2002; Linares et al., 2009; Hereş et al., 2012; Ruiz-Benito et al., 2013; Gea-Izquierdo et al., 2014), but a link between climate change and widespread increase of oak mortality in Southwestern Spain has not been established.

Dendrochronological data provide useful information to investigate forest dynamics in relationship with the environment (Fritts, 1976; Schweingruber, 1996). The xylem of Mediterranean evergreen species has anatomical features that make it difficult to establish chronologies (Cherubini et al., 2003), yet dendrochronological studies of *Q. ilex* has become well-established especially in recent times (Zhang and Romane, 1991; Cherubini et al., 2003; Campelo et al., 2009; Gea-Izquierdo et al., 2009, 2011). These studies have demonstrated that *Q. ilex* ring formation is very sensitive to climate, indicating the suitability of this species for dendroecological investigations. On other species, tree rings have been used to investigate growth patterns in declining forests and dead trees (see Schweingruber, 1996), to model mortality risk (e.g. Bigler and Bugmann, 2004) and to find relationships between mortality processes and external factors (e.g. Pedersen, 1998; Camarero et al., 2003; Bigler et al., 2006). Moreover, annual tree-ring widths can be used to estimate the basal area increment (BAI), as an indicator of forest productivity (Piovesan et al., 2008; Di Filippo et al., 2010). The inverse relationship between ring width and age is eliminated when radial growth is calculated as BAI (Biondi and Qeadan, 2008). In the absence of major ecological constraints, changes in BAI should be positive or approach an asymptote in adult trees (Poage and Tappeiner, 2002; Biondi and Qeadan, 2008; Sillett et al., 2010). Thus, recent studies have interpreted negative BAI trends in adult trees as evidence that trees have entered a declining phase (Piovesan et al., 2008; Di Filippo et al., 2010; Gea-Izquierdo et al., 2014).

In this paper, our objective was to assess the role of climate change on the widespread mortality of holm oaks occurring in the SW Iberian Peninsula. We used century-long tree ring chronologies of dead holm oaks (*Q. ilex* ssp. Ballota [Desf.] Samp.) from two managed open woodlands. We examined the climate change-related growth variability to verify whether negative impacts of climate change were reflected in the growth patterns before tree death. We analyzed (i) the relationship between climate and growth to determine the climatic variables that mostly influenced tree growth, (ii) the shifts in climate-growth relationships over time to evaluate the sensitivity of trees to the changing climate, and (iii) the low-frequency growth variability to identify its connections with climate trends.

2. Materials and methods

2.1. Study sites

The study sites are located in the province of Huelva, Spain (Fig. 1). Oak samples were collected from two monitoring experimental plots (2.9 ha per plot): Calañas (CA, 37° 31′ N; 6° 55′ W; 165 m a. s. l.) and Huerto Ramirez (HR, 3° 34′ N; 7° 20′ W; 200 m a. s. l.). The two stands are representative of oak open-woodlands in the SW Iberian Peninsula that are primarily used for livestock management. In the two stands the understory layer is composed of *Cistus ladanifer, C. crispus, C. monspeliensis* and an herbaceous layer of grasses. Soils in CA are shallower Regosols, Leptosols and Cambisols (25 to 50 cm depth) and deeper soils in HR range from Regosols and Cambisols (40–70 cm depth) to Acrisols, Alisols and Lixisols (60–100 cm depth). The stand density was 54 trees ha^{-1} (basal area: 4.5 m^{2} ha^{-1}) till 2006 in CA and 74 trees ha^{-1} (basal area: 5.2 m^{2} ha^{-1}) till 2010 in HR. Dead trees were logged since 2007 in CA and since 2011 in HR, and this led to decreases of stand density (Supplementary material 1). Similar to holm oak open woodlands in the province of Huelva, both stands present canopy dieback, widespread mortality and no regeneration.

2.2. Inventory of defoliation and mortality

Tree inventories were performed in the two plots to characterize canopy defoliation and tree mortality. Tree defoliation, defined as the loss (i.e. fall or complete dryness) of leaves, twigs, and side branches, was monitored between November and December in 2010 at HR and in 2001, 2005, and 2006 in CA; additional observations were performed in both plots in June 2013. All living trees were classified as “slightly affected” when defoliation was 10–20% of the crown, partially affected when defoliation was 30–60%, heavily affected when canopy defoliation was greater than 60% (López and Sánchez, 2011). A tree was classified as healthy when the crown was undamaged or defoliation was less than 10%. Furthermore, tree inventories reported the decrease in stand density due to the logging of dead trees. A tree was assumed to be dead when it was completely defoliated for at least two successive growing seasons.

2.3. Sampling and dendrochronological analyses

Complete cross-sections are needed for dendrochronological analysis of *Q. ilex* due to the complex xylem anatomy of this species (narrow rings, missing rings, and intra-annual density fluctuations (Cherubini et al., 2003; Campelo et al., 2007, 2009; Gea-Izquierdo et al., 2009)). Previous dendroecological studies on tree mortality in Iberian forests included tree ring data from dead and healthy trees for comparative analyses (e.g. Hereş et al., 2012; Gea-Izquierdo et al., 2014). In both sites of this study, the whole stand had signs of decline, thus establishing a chronology from healthy trees was not possible. Furthermore, obtaining stem sections of living *Q. ilex* trees is difficult because this species is protected in the region. Therefore, only cross-sections from dead trees were used for this study.

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