



Different growth strategies to invade undisturbed plant communities by *Acacia dealbata* Link



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ABSTRACT

The leguminous *Acacia dealbata* is one of the most aggressive invasive tree species worldwide. The invasion process of *A. dealbata* is mainly favoured by environmental disturbances. However, this species is currently invading unaltered plant communities and environmental factors influencing its spread in these ecosystems remain unclear. In this study, we evaluated the *A. dealbata* growth related to environmental factors in non-altered shrublands and pine forests in North-Western of the Iberian Peninsula. Permanent sampling plots were placed along the invasion edge in each plant community and height and stem diameter increments were recorded on labelled *A. dealbata* individuals throughout two years. The number of new *A. dealbata* saplings was recorded at the end of the study. Environmental factors such as physicochemical soil properties, native vegetation cover and plot features were evaluated in each sampling plot in order to determinate their effect on the *A. dealbata* growth. Our results showed a different growth pattern of *A. dealbata* depending on the studied plant community. Pine forests had *A. dealbata* individuals with significantly higher height and diameter increments and higher number of new *A. dealbata* individuals than shrublands. Additionally, results also showed a clear seasonality effect on *A. dealbata* growth that was stimulated in spring in pine forests, while it did not variate along the year in shrublands. Regarding environmental parameters influencing the growth of *A. dealbata*, we found a negative effect by the photosynthetically active radiation and plot orientation, but also a positive effect by soil potassium content and plot elevation. However, our results indicated that other environmental factors not evaluated here might be also influencing. We conclude that *A. dealbata* can invade unaltered areas and the intensity of the invasion depends on the ecosystem. The natural shrubland could be an effective barrier to slow down the rapid invasion of *A. dealbata*. These results contribute to new knowledge of *A. dealbata* colonization without help of environmental disturbances in the field, which may help to define hot-priority areas and establish new control strategies. Thus, we suggest that the management of *A. dealbata* in pine forests should receive preferential treatment on South-Western Europe.

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1. Introduction

Invasive alien species are one of the most important direct threats to biodiversity (Butchart et al., 2010; Vilà et al., 2011). The continuing increase of human mobility around the world acts as a key factor in the processes of biological invasions, facilitating migration and colonization of species (van Kleunen et al., 2015). Considered globally, invasions by exotic trees are a pervasive component of the global change that severely impact on ecosystems they invade and are able to spread quickly worldwide (Richardson et al., 2013; Richardson and Rejmánek, 2011;

Simberloff et al., 2013). Ecosystem transformations and homogeneity caused by invasive species result in loss of native species and altered ecosystem processes and services fundamental to resident organisms including humans (Hierro et al., 2005; Pejchar and Mooney, 2009; Wardle et al., 2011).

One of the most problematic exotic group worldwide consists of woody Australian acacias belonging to subgenus Phyllodineae (Lorenzo et al., 2010a; Lorenzo and Rodríguez-Echeverría, 2015; Richardson and Rejmánek, 2011). In fact, increasing literature reporting above and belowground impacts of these invaders has been released during the last years (e.g. Fuentes-Ramírez et al., 2011; Lorenzo et al., 2012; Rodríguez-Echeverría et al., 2013; Lazzaro et al., 2014; Aguilera et al., 2015a,b; Marchante et al., 2015; Guisande-Collazo et al., 2016). Among the invasive Australian acacias, *Acacia dealbata* Link is the third most widespread species of

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exotic Australian acacias around the world (Richardson and Rejmánek, 2011), being Europe particularly affected by the invasion of this species (Lamarque et al., 2011; Lazzaro et al., 2014; Lorenzo et al., 2016, 2012; Lorenzo and Rodríguez-Echeverría, 2012).

Invasion by *A. dealbata* lead to reduced light at understory level (Lorenzo et al., 2016), low availability of soil water (Le Maitre et al., 2011), increased content of soil nutrients (N, NH_4^+ , NO_3^- , C, P) and soil acidification (González-Muñoz et al., 2012; Lazzaro et al., 2014; Lorenzo et al., 2010b; Lorenzo and Rodríguez-Echeverría, 2012), altered soil microbial communities and enzymatic activities (Lazzaro et al., 2014; Lorenzo et al., 2013, 2010b, Rodríguez-Echeverría et al., 2013, 2012; Souza-Alonso et al., 2015, 2014), and diminished native plant cover and diversity under its understory (Fuentes-Ramírez et al., 2010; González-Muñoz et al., 2012; Lazzaro et al., 2014; Lorenzo et al., 2012).

The invasion success of *A. dealbata* is mainly related to soil perturbations and fire that promotes seed germination (de la Cueva, 2014; Hernández et al., 2014; Lorenzo et al., 2010a), high light availability (González-Muñoz et al., 2011), drought resistance (González-Muñoz et al., 2011; Maslin, 2001), soil previously modified by itself (Lorenzo and Rodríguez-Echeverría, 2012; Rodríguez-Echeverría et al., 2013) and the release of natural allelopathic compounds that interfere with normal process of plant and soil microbes (Aguilera et al., 2015a; Lorenzo et al., 2013, 2011, 2008). Interestingly, a recent work conducted under field conditions found that microhabitat and soil modifications by *A. dealbata* are more relevant than allelopathy during the invasion process of this species (Lorenzo et al., 2016).

Despite the ability of *A. dealbata* to preferentially invade altered ecosystems both by external perturbations or by its own-induced changes, this species can also establish and spread in plant communities where no apparent disturbances were observed (González-Muñoz et al., 2012; Lorenzo et al., 2010b; Lorenzo and Rodríguez-Echeverría, 2012). The problems associated to invasion of *A. dealbata* might be acute in South-Western Europe since the invaded area by *A. dealbata* is expected to expand in this region (Hernández et al., 2014).

Environmental disturbances clearly facilitate the spread of *A. dealbata* (de la Cueva, 2014; Hernández et al., 2014). However, environmental factors favouring the colonization of unaltered ecosystems by this invader have received little attention. The present study aimed at determining the growth strategies of *A. dealbata* in different unaltered plant communities and biotic and abiotic factors influencing them. The specific objectives were: (1) to assess the two-year and seasonal growth patterns and the recruitment of new individuals of *A. dealbata* in non-disturbed shrublands and pine forests in the potential-expanded area predicted by Hernández et al. (2014), and (2) to evaluate whether vegetation features, soil physicochemical properties and micro-physiographic characteristics are influencing the growth of this invasive species. We hypothesize that *A. dealbata* will grow in both studied plant communities, as this invader can establish in different ecosystems (Lorenzo et al., 2010a,b; Lorenzo and Rodríguez-Echeverría, 2012). However, we hypothesize that its performance will vary depending on local environmental factors. The results from this study may assist practitioners to make decisions in mitigating invasive *A. dealbata* further spread in forest ecosystems.

2. Materials and methods

2.1. Study area and sampling plots

The study was conducted in four shrublands and four *Pinus pinaster* forests located in North-Western of the Iberian Peninsula (Ourense, Galicia, Spain) which have been invaded by *Acacia dealbata* for

30 years (Table 1). All plant communities were located on granitic rock (Taboada, 1992), had the same ombrothermic domain-warm temperate climate and dry summer (Martínez and Pérez, 1999) (Table S1), and have not been recently disturbed. The average annual precipitation was different between sampling years, with the precipitation in Year 1 being almost double compared to Year 2 and the average of the 1972–2000 period (Table S1). The native shrublands were dominated by Fabaceae shrubs such as: *Cytisus striatus* (Hill) Rothm, *Pterospartum tridentatum* (L.) Willk. and *Ulex gallii* subsp. *breoganii* (Castrov. & Valdés Berm.) Rivas Mart. & al., and some Cistaceae, ferns and herbs. Pine forests were dominated by *Pinus pinaster* Aiton with an understory of Fabaceae and Cistaceae shrubs, ferns and herbs (Table 1). A sampling plot of 20 m × 5 m was placed along the invasion edge in each plant community (Fig. 1). The invasion edge was defined as the imaginary line between the *A. dealbata* patch (with *A. dealbata* density higher than five individuals per square meter) and the studied plant community with scattered individuals of *A. dealbata*. Within each sampling plot, four permanent 2 m × 5 m subplots containing at least five individuals of *A. dealbata* were randomly distributed (Fig. 1).

2.2. Monitoring of *Acacia dealbata* growth

Within each subplot, we randomly selected and labelled five individuals of *A. dealbata* to assess growth parameters. The growth parameters monitored for each individual were height and stem diameter. Plant height was the distance between ground and the highest photosynthetic tissue in the canopy. The stem diameter was measured at 10 cm from the soil layer (Cornelissen et al., 2003). The *A. dealbata* growth was periodically monitored for two years, from June 2013 to June 2015; with a total of seven samplings dates throughout the study (Fig. 1). Two-year increment in height (HI) and two-year increment in stem diameter (SDI) were determined for each individual by calculating the difference between the value recorded in June 2015 and the initial value registered in June 2013. Additionally, spring, summer and winter increments in height and stem diameter were determined by calculating the difference between sampling dates for each seasonal period. We also registered all new individuals of *A. dealbata* which appeared after two years in each permanent subplot.

2.3. Determination of abiotic and biotic environmental parameters

We determined soil physicochemical properties, vegetation features and physiographic parameters for each subplot. Three soil samples (0.04 m² and 10 cm deep) were randomly collected in each subplot in June 2013. Soil samples from the same plot were pooled together, air-dried and sieved through a 2-mm mesh. Total carbon (C), total nitrogen (N), phosphorous (P), potassium (K), soil moisture and pH were analysed following Lorenzo et al. (2010c) and Soil Survey Staff (2010). The vegetation cover was recorded in three random quadrats (0.5 m × 0.5 m) and estimated as the percentage of the area covered by all vascular and non-vascular plant species in the quadrat (Lorenzo et al., 2012). The orientation (established by 16-wind compass rose) and elevation of each subplot were also determined (Table 1). The photosynthetically active radiation (PAR) was recorded for each pine forest and shrubland at two meters from the soil in five points at least 15 m apart from each other. The PAR was measured in a sunny day in autumn when there was a lower variation in the irradiance levels among the studied plots.

2.4. Data analysis

The effect of plant community (shrublands and pine forests) on HI and SDI was assessed using Linear Mixed Models (LMMs) via

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