

Case studies of the expansion of *Acacia dealbata* in the valley of the river Miño (Galicia, Spain)

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Abstract

Aim of study: *Acacia dealbata* is a naturalized tree of invasive behaviour that has expanded from small plots associated with vineyards into forest ecosystems. Our main objective is to find evidence to support the notion that disturbances, particularly forest fires, are important driving factors in the current expansion of *A. dealbata*.

Area of study: We mapped its current distribution using three study areas and assesses the temporal changes registered in forest cover in these areas of the valley of the river Miño.

Material and methods: The analyses were based on visual interpretation of aerial photographs taken in 1985 and 2003 of three 1 × 1 km study areas and field works.

Main result: A 62.4%, 48.6% and 22.2% of the surface area was covered by *A. dealbata* in 2003 in pure or mixed stands. Furthermore, areas composed exclusively of *A. dealbata* make up 33.8%, 15.2% and 5.7% of the stands. The transition matrix analyses between the two dates support our hypothesis that the areas currently covered by *A. dealbata* make up a greater proportion of the forest area previously classified as unwooded or open forest than those without *A. dealbata* cover. Both of these surface types are the result of an important impact of fire in the region. Within each area, *A. dealbata* is mainly located on steeper terrain, which is more affected by fires.

Research highlights: *A. dealbata* is becoming the dominant tree species over large areas and the invasion of this species gives rise to monospecific stands, which may have important implications for future fire regimes.

Key words: fire regime; mimosa; plant invasion; silver wattle.

Introduction

Acacia dealbata Link. is one of several *Acacia* tree species originated from Australia and Tasmania that has naturalised in western regions of the Iberian Peninsula. The presence of *A. dealbata* in Galicia (Spain) stems from its use in certain public works, such as the railway at the beginning of the 20th century (García, 1979), and above all, to its subsequent use in vineyards, which commonly form part of the landscape in the area studied in this paper. In some areas, small plots of *A. dealbata* were cultivated as a source of posts. These plots were harvested in short rotations to obtain posts from the sprouts. This cultural practice has led to the presence of disseminated populations of *A. dealbata* throughout the area in question. *A. dealbata* has invaded adjacent forest areas from these initial sources as a result of the complex interrelation between cultivated and forested areas in Galicia.

The invasive nature of *A. dealbata* has been documented in several places such as North Western Spain (Carballeira and Reigosa, 1999; Sanz-Elorza *et al.*, 2001, 2004; Lorenzo *et al.*, 2010), Chile (Fuentes-Ramírez *et al.*, 2011) or South Africa (de Neergaard *et al.*, 2005). In Portugal, geographically proximate to our study areas, the invasive behaviour of *A. dealbata* (Aguar *et al.*, 2001), as well as other *Acacia* species such as *A. melanoxylon* R.Br. (Santos *et al.*, 2006) or *A. longifolia* Andrews in dune ecosystems on the Atlantic coast (Marchante *et al.*, 2004) has also been studied.

The expansion of *A. dealbata* populations is favoured by several characteristics which make this species a successful invader. It has a high capacity for regeneration after disturbances by means of vegetative (Sheppard *et al.*, 2006) or sexual reproduction (González-Muñoz *et al.*, 2011). Therefore, the increased presence of this species is favoured by gaps in the vegetation due to disturbances such as forest fires (Ough, 2001) or clearings for the provision of infrastructures (Aguar *et al.*, 2001; Spooner *et al.*, 2004). It also pos-

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sesses a high degree of phenotypic plasticity, which facilitates its adaptation to differing environmental conditions. Finally, the allelopathic effects of exudates of *A. dealbata* on other plants are well documented (Casal *et al.*, 1985; Reigosa *et al.*, 1998; Carballera and Reigosa, 1999; Lorenzo *et al.*, 2011).

The presence of *A. dealbata* produces wide ranging impacts on ecosystems, which increase both with time and level of disturbance. The result is the transformation of ecosystems and the alteration and reduction of ecosystem service delivery (Le Maitre *et al.*, 2011). *A. dealbata* stands accumulate a massive seed bank, which enables them to become dominant following disturbances. This positive relationship with disturbances is considered to be one of the most important factors behind the invasive spread of the species in Mediterranean areas (Brooks *et al.*, 2004; Lorenzo *et al.*, 2010; Le Maitre *et al.*, 2011).

Forest fires are the main disturbance factor in forest ecosystems in large areas of Spain, including Galicia (Moreno *et al.*, 1998; de la Cueva *et al.*, 2006). Additionally, as in other cultural landscapes, the vegetation cover has suffered major alterations. Certain species, such as *Eucalyptus globulus* or *Pinus pinaster*, have either been introduced by man or their distribution areas have been modified by human activity (Bellot, 1978; Rivas-Martínez, 1987; Blanco *et al.*, 1997). Furthermore, a large proportion of the current forested area has been cultivated in former periods thus making it more susceptible to invasion by non-native plants (Brooks *et al.*, 2004; Rejmánek *et al.*, 2005; Von Holle and Motzkin, 2007).

Despite the importance that *A. dealbata* has acquired in numerous parts of the western Iberian Peninsula, the scientific information available at present in relation to its status in Spain remains incomplete. Hence, for example, scarce research has been undertaken with regard to the patterns of territorial expansion (*e.g.* Vilà *et al.*, 2006; Lorenzo *et al.*, 2010). This lack of scientific documentation on the mechanisms and processes associated with the expansion of *A. dealbata* at landscape level contrasts with the concern expressed in environmental, scientific and administrative media given the increasing presence of this species in many areas. This concern is evidenced by the various eradication initiatives which are already being considered on a regional scale by the Galician Regional Government (Xunta de Galicia, 2007).

Focusing on three study areas, this paper attempts to document the temporal dynamics of three land-

scapes in the valley of the river Miño where *A. dealbata* has now become a relevant tree species. The main objectives are (1) to document the current spatial distribution and (2) to analyse the temporal dynamics of the forest cover in the three study areas. For this purpose, the forest cover at the three 1 km × 1 km sites was mapped in accordance with aerial photographs taken in 1985 and 2003. The intention was to quantify the area in which *A. dealbata* was present in 2003 and to document the changes in forest cover between 1985 and 2003. Unfortunately, it was not possible to infer the presence of *A. dealbata* from the photographs taken in 1985. Through the approach used in this paper to interpret the forested landscape, our intention is to verify the notion that disturbances, particularly forest fires, have been and continue to be determinant in the expansion of *A. dealbata*. Our working hypothesis is that zones currently covered by *A. dealbata* originated (in a greater proportion than those areas not covered) from unwooded or open forest areas as a result of previous disturbances.

Material and methods

Study area

The area under study is located in the valley of the river Avia, close to where it flows into the river Miño, in the city of Ribadavia (administrative province of Ourense, Galicia) (Figs. 1a and b). As regards the potential natural vegetation, this area is located in an extensive ecotone between Mediterranean formations to the south (with sclerophyllous tree species such as *Quercus ilex* L. and *Q. suber* L. and deciduous as *Q. pyrenaica* Willd.) and more temperate deciduous forest to the north (with tree species like *Q. robur* L.) (Bellot, 1978; Rivas-Martínez, 1987). The present vegetation cover comprises a mosaic of small polygons dedicated to crops (mainly vineyards) within a forest matrix. The present forest vegetation is dominated by species of rapid growth such as *Pinus pinaster* Ait. and *Eucalyptus globulus* Labill. *P. pinaster* is a native species frequently used in forest plantations. The mean annual temperature and the mean annual rainfall were 14.5°C and 817 mm respectively for the period 1972-2000, according to the meteorological station at Ourense (143 m a.s.l and 20 km from our study area). This is one of the regions in Spain with the greatest potential forest productivity, with values of over 8 m³

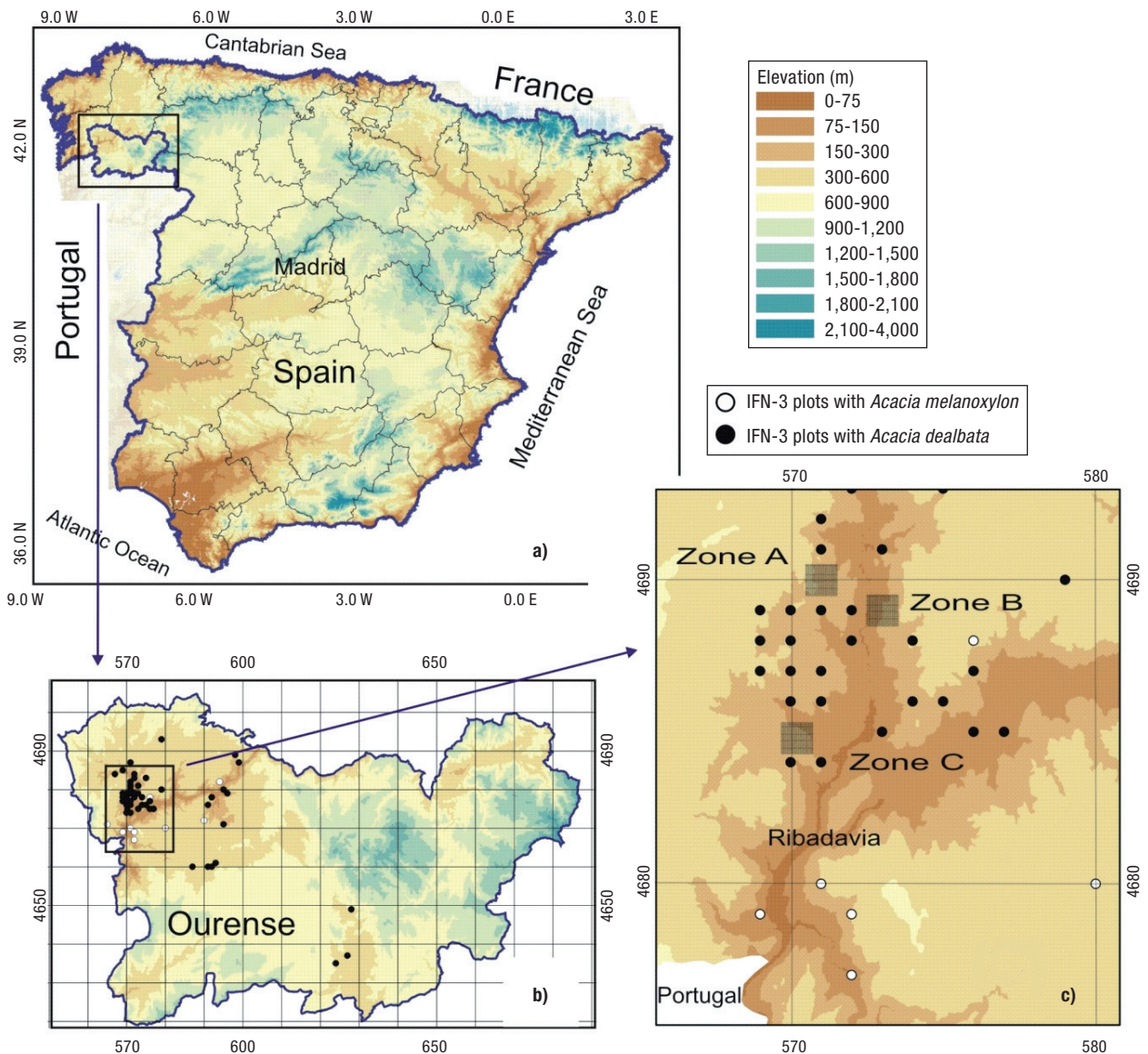


Figure 1. Maps showing the study area. (a) Boundaries of peninsular Spain with elevations and administrative boundaries (provinces) where the province of Ourense is highlighted. Galicia is an autonomous community formed by the four north-western provinces of Spain. (b) Province of Ourense with the IFN-3 plots where *Acacia* species were detected. A 10 × 10 km grid used as reference for fire incidence has been superimposed based on the UTM projection (Zone 29, Datum European 1950). (c) Location of the three 1 × 1 km zones under study (a, b and c) in the river Avia valley close to where this river flows into the river Miño in Ribadavia. The IFN-3 plots (black for *A. dealbata* and white for *A. melanoxylon*) and the 10 × 10 km grid are also superimposed.

ha⁻¹ year⁻¹ (Sánchez-Palomares and Sánchez-Serrano, 2000).

The three 1 × 1 km study areas were selected based on data from the third National Forest Inventory (IFN-3) indicating the presence of *Acacia* species in the plots (Vallejo and Villanueva, 2002). The two *Acacia* species identified were present in 55 of the 1480 plots surveyed in the IFN-3 in the province of Ourense in the summer of 1998. *A. dealbata* was present in 46 plots

(3.1%) and *A. melanoxylon* in 9 (0.6%) of them (Fig. 1b). The IFN-3 involves periodic sampling of plots on a 1 × 1 km grid in forested areas. Plots are circular with a maximum radius of 25 m. Fig. 1c shows the IFN-3 plots in which the two *Acacia* species were present along with the location of the three study areas interpreted from aerial photography. Unfortunately, in the IFN-2 performed 10 years previously, the *Acacia* species were not codified and therefore the data

from this inventory is of little use for the purposes of this study.

Forest fire incidence in this area was investigated using the statistical information provided by the Spanish forest administration. Each fire was referenced to the 10 × 10 km grid-unit (based on the UTM projection) in which the fire started. The fires considered are those which resulted in a burned forest area ≥ 0.1 ha. Based on this information combined with land-cover information (NATLAN, 2000), fire frequency (number of fires registered in 10,000 ha of forest area and year) and the fire rotation period (number of years required to burn an area equivalent to the reference surface) were calculated for the period 1974-2005 (32 yrs) in the four 10 × 10 km grid-units in which the three study areas are located.

Aerial photograph interpretation and analysis procedure

Aerial photographs taken in the summers of 2003 and 1985 were used to perform a detailed visual classification of the land cover in the three study areas. Interpretation of the images was carried out using a computer display at a very high (approximately 1:2000) resolution. We focused on forest cover type and composition, delimiting areas of homogeneous tree cover and physiognomy. The interpretation of the territory from the photographs was backed up by extensive field work (year 2007) to guarantee the accuracy of the visual classification.

The colour photographs taken in 2003 were already ortho-projected and had a spatial resolution of 25 cm. These images allowed the forest cover classification to be carried out and to identify the patches and forest stands covered by *A. dealbata*. The 1985 photographs were in black and white at a scale of 1:25000. The latter were scanned and geometrically adjusted to the 2003 images by means of an ortho-correction process to guarantee an acceptable superposition between the two land cover classifications. The ortho-correction includes the definition of an internal model based on the fiducial marks of the photogram and on the focal distance of the camera, along with an external model based on ground control points that are common to the two images. A digital terrain model is also necessary to minimise distortions due to the terrain. The errors (measured by the root mean square error) obtained in the three photograms from 1985 had values of approximately 10 m. The spatial resolution of these images was set at 50 cm. The forest cover classification for 1985 was carried out from the images but it was not possible to identify the areas covered by *A. dealbata*. Various features of the geographic information systems MiraMon (Pons, 2002) and ArcGIS (ESRI, 2005) were used in the processing and interpretation of the images.

Visual classification of the land cover and the presence of *A. dealbata* was based on a relatively simple legend (Table 1) in which two forest cover attributes were contemplated: (1) canopy crown closure in 1985 and 2003 and (2) forest stand type in 2003 only. The Canopy Crown Closure (CCC) of the trees or large

Table 1. Land cover classes identified for the year 2003 in the three 1 × 1 km study areas (zones A, B and C) by means of the photographic interpretation conducted. The forest area was classified based on two forest cover attributes: canopy crown closure (1) and forest stand type (2). Classes in which *A. dealbata* is present are shown in bold. The values in each column are expressed in percentages. The number of points used in calculations are in brackets

Land cover attributes photo-interpreted			<i>A. dealbata</i> areas in the year 2003		
Type of use	Canopy crown closure (1)	Forest stand type (2)	Zone A	Zone B	Zone C
Forest area	Closed forest (1.1)	<i>A. dealbata</i> (2.1)	(132) 29.9	(21) 4.8	(15) 3.4
		Mixed <i>A. dealbata</i> (2.2)	(53) 12	(29) 6.6	(4) 0.9
		Other species (2.3)	(13) 2.9	(73) 16.6	(73) 16.6
	Open forest (1.2)	<i>A. dealbata</i> (2.1)	(17) 3.9	(46) 10.4	(10) 2.3
		Mixed <i>A. dealbata</i> (2.2)	(73) 16.6	(118) 26.8	(69) 15.6
		Other species (2.3)	(22) 5	(39) 8.8	(86) 19.5
	Unwooded (1.3)		(34) 7.7	(30) 6.8	(130) 29.5
			(68) 15.4	(46) 10.4	(29) 6.6
	Cultivated		(29) 6.6	(39) 8.8	(25) 5.6
Urban/water			(441) 100	(441) 100	(441) 100
Total					

shrubs was quantified into three classes: (1.1) closed forest areas with a CCC of more than 60%, (1.2) open forest areas with a CCC between 20% and 60% and (1.3) unwooded forest areas or areas with scattered trees with a CCC of less than 20%. The forest stand types distinguished: (2.1) stands dominated by *A. dealbata*, (2.2) stands with a mixture of *A. dealbata* and other forest trees and (2.3) polygons occupied by other forest species. The identification of *A. dealbata* stands, that is, the forest stand type classification, was only performed for the year 2003 and for patches with a CCC greater than 20% (1.1 closed forest and 1.2 open forest), as can be seen in Table 1. Finally, the cultivated areas (non-specified although most of them were vineyards) as well as areas with no vegetation cover (urban surface areas, infrastructures and water bodies) were identified and mapped.

The analyses of forest cover changes were based on the confusion matrix between both dates for each zone (study area). A sample of 441 points derived from a 50 × 50 m grid overlaid on the polygons delimited in each of the 1 × 1 km study areas was used (Fig. 2). In addition we performed non-parametric rank correlations (Spearman r_s) as a measure of association between both dates. In this analysis, the values of each of the variables are ranked from smallest to largest, and the Pearson correlation coefficient is computed on the ranks. The procedure for obtaining quantitative values involved counting the number of cells in each condition in 3 × 3 windows for each study area and year. In this way, we obtained quantitative values for 49 “samples” in each zone. For each zone, the three types of canopy crown closure in 1985 (unwooded, open, and closed) were correlated with the presence (in pure or mixed stands) and the absence of *A. dealbata* in 2003. Finally, to evaluate the presence of statistically significant differences in the topographical attributes between areas covered (or not) by *A. dealbata*, we used the non-parametric range-based Mann-Whitney U test. We assessed the differences in three topographical attributes (altitude, aspect and slope) using the 441 sampling points available in each zone.

Results

Areas covered by *A. dealbata* in 2003

According to the interpretation procedure followed, forested areas dominated the three study areas with

proportions ranging from 78% in zone A to 87.8% in zone C. The proportion of the total surface area in which *A. dealbata* was present in 2003 was 62.4% in zone A, 48.6% in zone B and 22.2% in zone C (Table 1 and Fig. 2). Fig. 2 shows the canopy crown closure for 1985 on the left (Fig. 2a, c and e), while on the right, Fig. 2b, d and f illustrates the canopy crown closure and forest stand type in 2003, indicating the presence of *A. dealbata* in the three zones.

The analysis carried out reveals that *A. dealbata* dominates zone A, giving rise to an almost continuous patch of closed forest (41.9% of the total surface) comprised mainly of stands dominated exclusively by *A. dealbata* (33.8%) (Fig. 2b and Table 1). Although *A. dealbata* occupies almost half of the territory in zone B (Fig. 2d and Table 1), its presence is less significant given that it is mainly found in open forest areas (37.2%) and in mixed forest stands (33.4%). The presence of *A. dealbata* in zone C is lower than in the other two (Fig. 2f and Table 1). In this case, *A. dealbata* is mainly located in open forest areas (17.9%) and in stands in which *A. dealbata* is mixed with other species (16.5%).

Areas covered by *A. dealbata* in 2003 in relation to the canopy crown closure in 1985

Most of the areas in which *A. dealbata* was detected in 2003 were already forested in 1985. The areas classified as “non-forest” in 1985 do not contribute significantly to the present area covered by *A. dealbata*. The areas classified as cultivated in 1985, but currently covered by *A. dealbata*, account for 2.1% in zone A, less than 1% in zone B, and a higher percentage in zone C, with mean values of 8.2% (Table 2).

In zone A, most of the area covered by *A. dealbata* in 2003 was classified as “open forest” in 1985 (52%), while the rest was classified as ‘unwooded’ (22.5%) or “closed forest” (22.5%). These proportions are similar to those of areas currently classified as open or closed forest areas covered by *A. dealbata* (Table 2). In zone B, 44.4% of the area covered by *A. dealbata* in 2003 was classified as open forest in 1985, while the rest was classified as unwooded forest (41.1%) and closed forest (13.6%). In this area, the proportions of what is presently classified as open or closed forest areas differ somewhat. A greater proportion of open forest area (with presence of *A. dealbata* in 2003) originates from areas classified as ‘closed forest’ in 1985

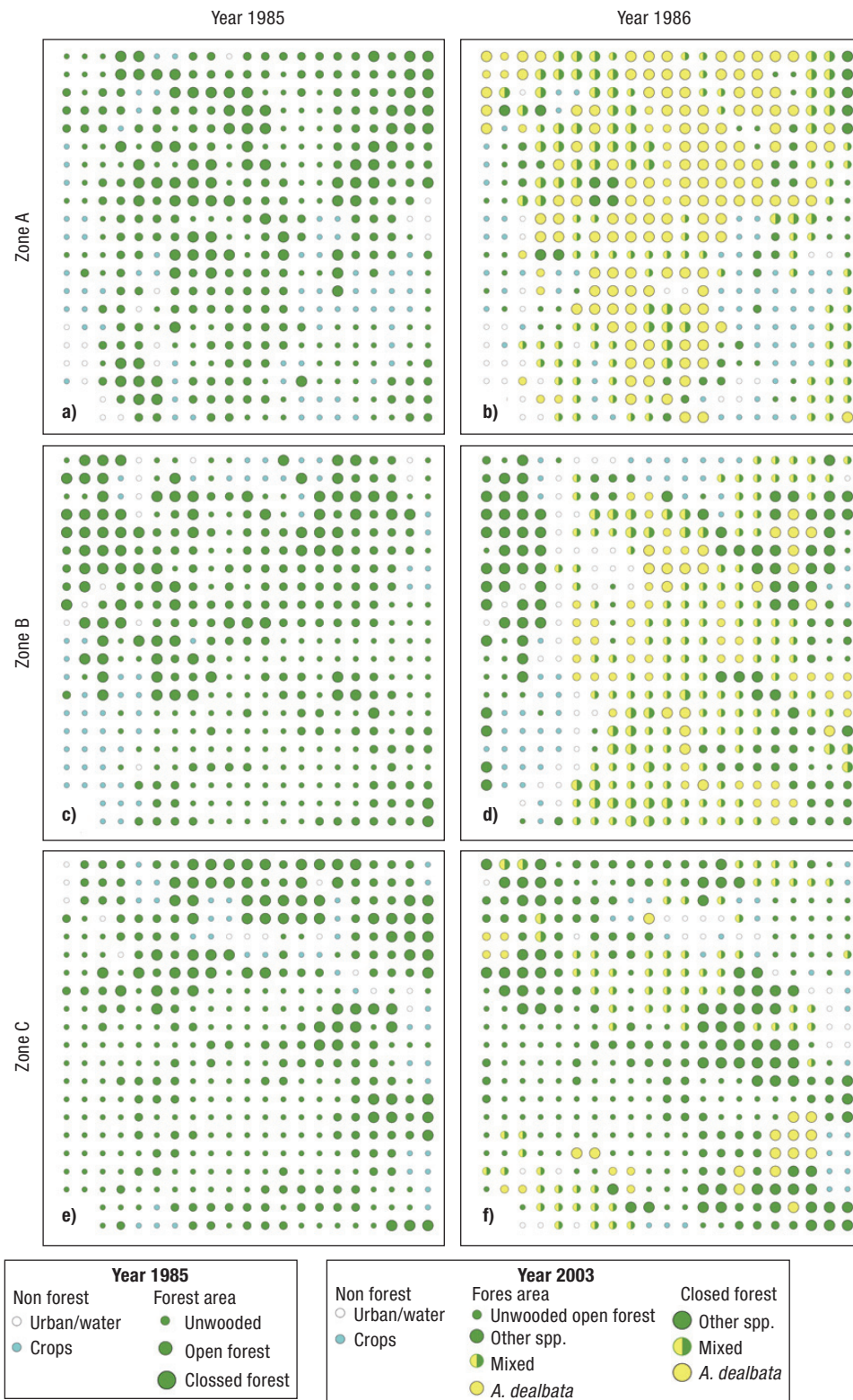


Figure 2. Land cover classification based on the interpretation of aerial photographs. Canopy crown closure (a, c, and e) in 1985, and canopy crown closure and forest stand type in 2003 (b, d and f) in the three study areas. Each image is formed by 441 points distanced by 50 m and covering the 1 × 1 km zone. The presence of *A. dealbata* (as the dominant tree or in mixed forest stands) has only been identified for the year 2003.

Table 2. Transition matrix, simplified, between the forest cover interpretations performed for the years 1985 and 2003. Canopy Crown Closure in 2003 (differentiating between open forest, closed forest and total) in areas covered by *A. dealbata* as a function of the canopy crown closure in 1985. The percentages in each column are adjusted to 100. In brackets in each cell, is the number of points involved in the calculations. The absolute values are those shown in Table 1

1985-2003	Areas covered by <i>A. dealbata</i> in the year 2003 (%)								
	Zone A			Zone B			Zone C		
	Canopy crown closure			Canopy crown closure			Canopy crown closure		
	Closed	Open	Total	Closed	Open	Total	Closed	Open	Total
Closed forest	(43) 23.2	(19) 21.1	22.5	(2) 4.0	(27) 16.5	13.6	(2) 10.5	(22) 27.8	24.5
Open forest	(98) 53.0	(45) 50.0	52.0	(26) 52.0	(69) 42.1	44.4	(13) 68.4	(16) 20.3	29.6
Unwooded forest	(40) 21.6	(22) 24.4	22.5	(22) 44.0	(66) 40.2	41.1	(3) 15.8	(34) 43.0	37.8
Cultivated	(3) 1.6	(3) 3.3	2.2	(0) 0.0	(1) 0.6	0.5	(1) 5.3	(7) 8.9	8.2
Urban/water	(1) 0.5	(1) 1.1	0.7	(0) 0.0	(1) 0.6	0.5	(0) 0.0	(0) 0.0	0.0
Total	(185) 100	(90) 100	100	(50) 100	(164) 100	100	(19) 100	(79) 100	100

(Table 2). In zone C, the areas covered by *A. dealbata* in 2003 were mainly preceded by unwooded areas (37.8%), followed by open forest areas (29.6%) and closed forest areas (24.5%). In this zone, the proportions varied in such a way that most of the areas presently occupied by *A. dealbata* in open forest areas originated from surfaces previously classified in 1985 as unwooded areas (43%), whereas those areas currently occupied by *A. dealbata* in closed forest originated from open forest areas (68.4%) (Table 2).

The results of the correlation analyses performed for the three types of canopy crown closure in 1985 (unwooded, open, and closed) with the presence (in pure or mixed stands) and the absence of *A. dealbata* in 2003 are shown in Table 3. It can be seen that the open and unwooded forest areas tend to show negative association with the absence of *A. dealbata*. Simi-

larly, the presence of *A. dealbata* in 2003 was found to be positively related to the amount of open or unwooded forest in all three zones. This pattern results in statistically significant correlations for the ‘unwooded’ category in Zone B ($r_s=0.57$) and values close to the significance level for the ‘open forest’ category in Zone A. The patterns obtained for “closed forest” are less clear (Table 3).

Topographic patterns in zones covered, or not, by *A. dealbata*

The topographic patterns of the forest areas (excluding those corresponding to cultivated and urban areas) which are either covered or not by *A. dealbata* in the three zones are illustrated in Fig. 3. The three box

Table 3. Correlation coefficients (r_s Spearman rank) between the three Canopy Crown Closure (CCC) conditions in 1985 and the absence and presence of *A. dealbata* (in pure or mixed stands) in 2003 for the three zones

Zone	1985 CCC	Absence of <i>A. dealbata</i> (2003)			Presence of <i>A. dealbata</i> (2003)		
		r_s	p	n	r_s	p	n
A	Closed forest	0.11	0.72	13	0.15	0.48	25
	Open forest	-0.05	0.84	19	0.28	0.08	40
	Unwooded forest	-0.23	0.39	16	0.02	0.93	28
B	Closed forest	0.32	0.14	23	0.07	0.77	20
	Open forest	-0.03	0.90	26	0.10	0.56	35
	Unwooded forest	0.20	0.38	22	0.57	0.00	32
C	Closed forest	0.23	0.30	22	-0.10	0.68	19
	Open forest	-0.18	0.27	38	0.14	0.48	29
	Unwooded forest	0.25	0.15	34	0.27	0.20	24

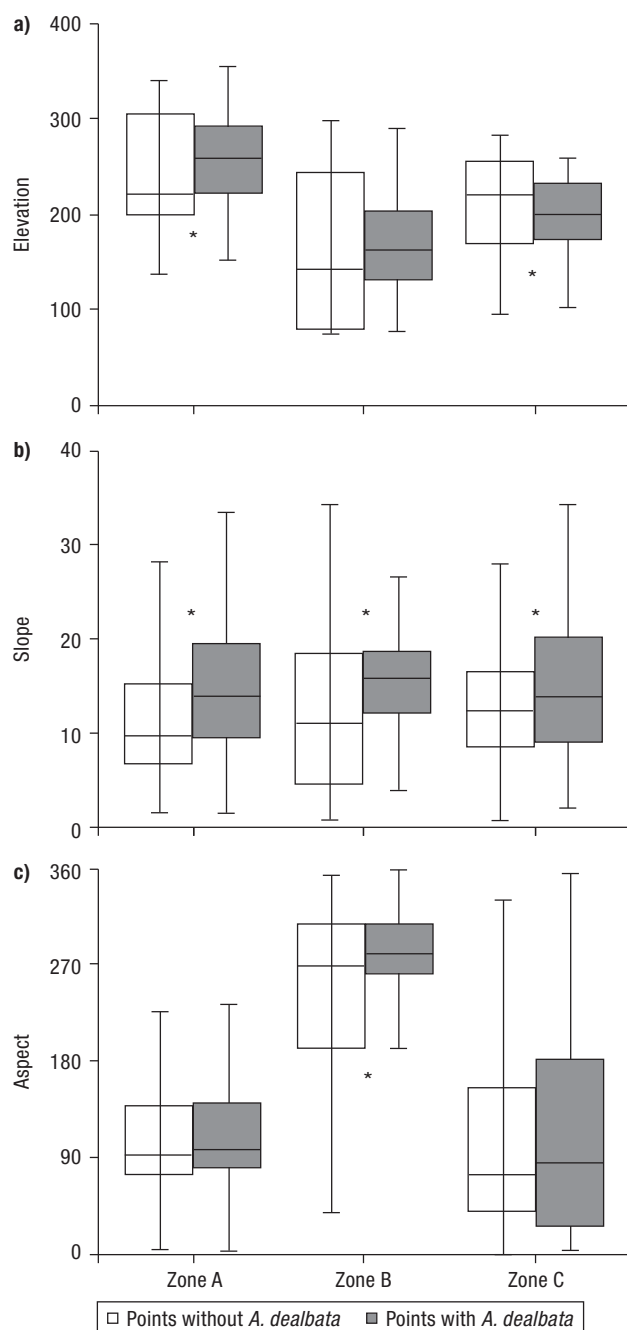


Figure 3. Boxplots showing the median, 50% of the values, and both the minimums and maximums once the outliers had been excluded for the three topographic variables: elevation (m), slope (degrees) and aspect (degrees) in the three zones (a, b and c). The shaded boxes correspond to forest areas with *A. dealbata* whereas the non-shaded areas are the areas in which it was not identified in 2003. The statistically significant differences (Mann-Whitney test) between both sets for each variable and each zone are shown with an asterisk ($p \leq 0.001$).

plots summarise the distribution of the data and show the results of the tests performed to compare the two

groups (areas with or without *A. dealbata* in 2003) in each zone. In zone A, the areas covered by *A. dealbata* were located at higher altitude than those not occupied by *A. dealbata*. In zone B, no significant differences were observed, whereas the areas not occupied by *A. dealbata* in zone C were found at a higher altitude (Fig. 3a). When comparing slopes, statistically significant differences were obtained in all three zones and a consistent pattern was identified: those forest areas in which *A. dealbata* was present had steeper slopes than areas not occupied by the species (Fig. 3b). Fig. 3c shows that plots A and C are predominantly east facing, whereas plot B faces west. In relation to the latter variable, statistically significant differences were only detected for zone B.

Forest fire incidence

The number of fires assigned to the four 10×10 km grid-units surrounding the study area for the period 1974-2005 ranged from 508 to 1658. These figures imply a mean of 31 forest fires per grid-unit per year. The forest area burned in the fires assigned to these grid-units ranged from 4639 to 13928 ha with a mean value of 9,952 ha. The fire frequency calculated ranged from 28 to 69 with a mean value of 42 fires $10,000 \text{ ha}^{-1} \text{ yr}^{-1}$. The fire rotation period ranged from 16 to 39 with a mean value of 26 years.

Discussion and conclusions

To our knowledge, this study represents the first attempt to make a detailed assessment at landscape level of the spatial impact of *A. dealbata* in the north-western region of the Iberian Peninsula. Despite the very high degree of cover identified in the three study areas (62.4%, 48.6% and 22.2%), quantifying *A. dealbata* cover at regional level would require a different sampling approach. In this study, we have focused on one of the regions in which *A. dealbata* is present, basing the selection of study areas on IFN-3 data for just one administrative province; in this case, Ourense. A general idea of the regional distribution of *A. dealbata* may be obtained from the IFN-3 data. It must be borne in mind that each of the analysed areas (1×1 km) is of a size equivalent to the area of influence of the IFN-3 plots (sampled on a 1×1 km grid) and that the three study areas used in this study are located among

these plots (see Fig. 1). Therefore, it follows that the situation observed in these study areas may be extrapolated to other locations. Unfortunately, it was not possible to accurately identify the presence of *A. dealbata* from the black and white photographs taken in 1985. In addition, the genera *Acacia* were not coded in previous forest inventories.

The results of the study confirm our working hypothesis that the areas currently occupied by *A. dealbata* generally originate from unwooded or open forest areas. The transition matrix between forest cover in 1985 and 2003 and the correlation analyses seem to support this hypothesis. In spite of the lack of statistical significance for some of the regression, the sign and the general tendencies support our hypothesis. In general, there is a positive relationship between the presence of *A. dealbata* in 2003 and the amount of open and unwooded forest in 1985. No edaphic or climatic restrictions exist in this region that hinder plant succession, either with or without human intervention, which gives rise to a closed forest in a relatively short period of time (Rivas-Martínez, 1987; Blanco *et al.*, 1997). This means that most unwooded or open forest areas originate from forest fires, highlighting the role of humans in facilitating the spread of organisms introduced through disturbance and landscape alteration (Sponner *et al.*, 2004; Ricciardi, 2007).

Expansion of *A. dealbata* and forest fires

An explicit study of the areas affected by forest fires was not undertaken, but the analysis of the statistical information relative to the region in which the three study areas are located underlines the huge impact that forest fires have on this region (Moreno *et al.*, 1998; de la Cueva *et al.*, 2006). In an analysis of the recent fire regime in several territories (ecozones) of peninsular Spain (de la Cueva *et al.*, 2012), it was found that the highest fire incidence was registered in the ecozone in which the study areas are located. Moreover, within this ecozone, fire incidence in our study area was higher than the average for the period 1974–2005. Median values in this ecozone, which comprises 373 10 × 10 km grid-units, indicate a fire frequency of 12.5 fires per 10,000 ha⁻¹ yr⁻¹ and a rotation period of 70 years. In the four grid-units surrounding the study area, the values obtained reflect much higher fire frequency (42 fires per 10000 ha⁻¹ yr⁻¹) and much shorter rotation periods (26 years). However, the high fo-

rest productivity in this region would enable a rapid regeneration of the plant cover, and therefore, very short recurrence intervals between fires (Vázquez *et al.*, 2002; Bond and Keeley, 2005). Furthermore, given that most fires in this area are not registered under extreme meteorological conditions, it is more likely that a proportion of trees will survive; giving rise to the type of forest that has been interpreted in this study as unwooded or open forest areas (Vázquez *et al.*, 2001).

The importance of forest fires on the present distribution of *A. dealbata* is supported not only by the confusion matrix analyses but also by the topographic pattern analyses performed. Significant differences were detected with regard to slopes. In the three zones studied, *A. dealbata* is located in areas with steeper slopes. This result may reflect the greater impact of fire in steeper areas where it spreads more easily (Rothermel, 1983; Burgan and Rothermel, 1984), and is often more difficult to combat (Vélez, 1990). Furthermore, areas with steeper slopes tend to be of a marginal nature where cultivation of crops may have been abandoned earlier (Lasanta, 1988), thus facilitating colonisation by *A. dealbata*. Similarly, such areas are often suitable candidates for occupation by this species due to the difficulties which they often present to forest management.

In a review of the genus *Acacia* as an invader, Lorenzo *et al.* (2010) grouped the biological attributes conferring invasiveness of *A. dealbata* into four major categories: a) capacity to take advantage of disturbances, particularly fire, b) phenotypic plasticity, c) vegetative reproduction and d) allelopathy. Several studies have dealt with the effect that exudates of *A. dealbata* have on the growth of other species, in which allelopathic effects have been described (Casal *et al.*, 1985; Reigosa *et al.*, 1998; Carballeira and Reigosa, 1999; Lorenzo *et al.*, 2011). These authors have revealed that the toxic effects of *A. dealbata* flowers influence the germination of other species. The main flowering season of *A. dealbata* occurs in February or March, which coincides with the germination of other species. Research into changes in soil properties has also revealed an increase in N concentration and organic matter in soils below *A. dealbata* due to the N fixation capacity associated with this species (May and Attiwill, 2003; Marcet *et al.*, 2006).

A. dealbata is highly resilient to forest fires and regenerates both through germination and sprouting from

roots and stems (Ough, 2001). Using a spatially explicit model of forest dynamics in a dryland forest landscape in Central Chile, Newton *et al.* (2011) conclude that the spread of *A. dealbata* only occurs in the presence of fire when combined with browsing and/or cutting. One of the ways in which the invasion of a new species may affect previously established ecosystems is through the modification of fuel properties, which can in turn affect fire behaviour and ultimately the fire regime characteristics (Brooks *et al.*, 2004). Grigulis *et al.* (2005) have documented the interaction between another invasive plant (a perennial grass that expands in areas of the Spanish Mediterranean coast) and the dynamics of the fuels associated with its expansion. These authors have suggested that short rotation periods favour the establishment of this grass, and in turn, that the presence of this species actually favours a potential increase in fire intensity due to fuel modification. In our study area, the results obtained along with the field observations appear to indicate that fires favour the presence of stands dominated by *A. dealbata*. However, it remains unclear as to whether the spread of fire is enhanced in stands of *A. dealbata* as opposed to the types of vegetation previously present. Given the large number of factors involved and the diversity of the communities in which *A. dealbata* is established, more data would be required to make a proper assessment of this aspect. However, the species is considered to be highly flammable in its natural distribution area, and its presence in close proximity to dwellings is not recommended in fire prone areas (Chladil and Sheridan, 2003).

In an interesting report concerning the impacts of invasive Australian *Acacia* in three case studies, Le Maitre *et al.* (2011) point to the fact that the three cases share some important drivers of change: the triggering of invasion by fire (or other disturbance), production of large, long-lived seed banks, increased soil nitrogen, suppression of native vegetation and eventual depletion of native seed banks as well as reduced restoration potential. All these aspects can be applied to our case. Additionally, however, in the case of *A. dealbata*, it is necessary to consider the post-fire resprouting capacity. Some authors (*e.g.* Lorenzo *et al.*, 2010) have already suggested that the spread of *A. dealbata* in Galicia may be assisted by human interference such as soil disturbance and fires. The results of the spatial analysis in this paper would appear to support this idea.

Conclusions

According to the conceptual model proposed by Richardson *et al.* (2000), *A. dealbata* may be considered not only as an invasive species in the sense that it is capable of expanding beyond the area into which it was introduced, but also a transforming species given that it modifies the characteristics of the ecosystem where it becomes established. In Galicia, *A. dealbata* fits into this category as we have seen in this work. The results of the study confirm our working hypothesis that the areas currently occupied by *A. dealbata* generally originate from unwooded or open forest areas, being these surfaces most of times the result of previous disturbances. In general, invasion by *A. dealbata* poses a threat to natural habitats through competition and replacement of native species, decreasing the native biodiversity and homogenizing the community. Despite its attractive blooms, the presence of this species is leading to a significant decrease in landscape diversity as well as substantial changes in the vegetation structure and composition. This study represents the first attempt to conduct a spatially explicit analysis of the interactions between landscapes, fire and species composition in Galicia. In the context of global change, it is expected that the importance of these interactions will increase in the future.

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