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Research article

Modelling *Acacia saligna* invasion in a large Mediterranean island using PAB factors: a tool for implementing the European legislation on invasive species

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Highlights

- SDM is an effective tool for predicting plant invasions
- An integrative PAB approach to explain *Acacia saligna* distribution in Sardinia
- Combined action of propagule pressure, abiotic, biotic factors promotes the invasion
- iSDM largely benefits from the use of high resolution and dedicated thematic layers
- iSDM is an effective tool for decision-making to prevent the invasion risk

Abstract

The present study aimed to investigate the role of propagule pressure (P), abiotic (A), and biotic (B) factors (collectively indicated as PAB) on the suitability of the Mediterranean island of Sardinia (Italy) to be invaded by the tree *Acacia saligna*, recently included in the list of invasive alien species of European Union concern.

To this aim, a binomial Generalized Linear Model was applied for disentangling the relationship between 432 *A. saligna* occurrence records and 10 thematic layers, at high-resolution (10 x10 m), used as proxies for the 3 categories of PAB variables. The 432 occurrence records of *A. saligna* were periodically monitored (period 2000-2018) to check the persistence of the populations and their invasive status. The predictive power of the model was evaluated by computing the mean of the AUC scores, through cross-fold validation. The model adequately described how the PAB factors influence the presence of *A. saligna* which is mainly shaped by abiotic factors such as topography, and biotic factors such as the presence of woody dune vegetation, and to a lesser extent by other predictors. The projection of the model to the whole island clearly shows that suitability varies at the landscape level due to the variation of the PAB across the territory. The probability of *A. saligna* occurrence near the coast is higher in sand dunes. In the internal areas of the island it occurs close to the roads and urban areas. This study and the tested methodology could represent a suitable tool to prioritize areas for the monitoring of *A. saligna* to meet the requirements of the Regulation (EU) No. 1143/2014 on Invasive Alien Species (the IAS Regulation).

Keywords Conservation planning, Generalized Linear Model, Invasive Alien Species Regulation, invasive Species Distribution Model, Sardinia.

1. Introduction

In 2002 the Conference of the Parties to the Convention on Biological Diversity (CBD) adopted the Guiding Principles on Invasive Alien Species (IAS Decision VI/23) as a basic policy response. The first CBD guiding principle states that prevention is generally far more cost-effective and environmentally desirable than measures taken after IAS introductions. Therefore, the identification of the major pathways of introduction and secondary spread, the areas and land uses more prone to invasion, and the implementation of early warning-rapid interventions are all key actions to be included in a national strategy for preventing IAS introduction, establishment and spread (Genovesi and Shine, 2004; Early et al., 2016). Predicting the risk from IAS establishment and negative impacts is of great importance for policy makers, land managers and other stakeholders, to delineate specific action plans and to choose or prioritize measures against IAS (Venette, 2015; Bazzichetto et al., 2018a). Therefore, invasive alien species distribution models and habitat suitability maps (iSDM) are a very useful tool producing reliable and repeatable information with which to inform decisions (e.g., Guisan and Thuiller, 2005; Broennimann and Guisan, 2008; Jiménez-Valverde et al., 2011; Petitpierre et al., 2012; Guisan et al., 2013). Nevertheless, application of iSDMs may have several limitations as a result of the invasion process, e.g. violation of the equilibrium assumption and underestimation the potential climatic niche of the species (e.g., Fournier et al., 2017; Barbet-Massin et al., 2018; Chapman et al., 2019).

The unified framework proposed by Blackburn et al. (2011) suggests that the invasion process can be divided into a series of stages from introduction to successful establishment until invasion. Many studies have addressed the interactions between alien species' invasive capacity and the susceptibility of habitats or communities to invasion (e.g., Pyšek and Richardson, 2008; Mathakutha et al., 2019). In their review of invasion ecology hypotheses, Catford et al. (2009) suggest considering each stage of invasion as a function of propagule pressure (P), abiotic environment (A) and biotic relationships (B) (PAB hypothesis). The propagule pressure, i.e. the number of introduced propagules, is a prerequisite for invasion (Colautti et al., 2006; Malavasi et al., 2014), while alien species establishment depends on the physical environment (abiotic filter; e.g., Malavasi et al., 2018) and on the biological features of the hosting community (biotic filter; e.g., Broennimann et al., 2012).

We decided to apply the PAB hypothesis to the well-known globally invasive plant *Acacia saligna*, an evergreen tree native to Western Australia (Maslin, 1974). It is a fast-growing tree that propagates both vegetatively and sexually, is well adapted to semiarid landscapes and quite resilient to fire (George et al., 2008). The current wide invasive range occupied by *A. saligna* is due to a combination of characteristics such as the adaptability to different environmental conditions, the large seed production and easy germination, the establishment of a rich seed-bank in the soil (Maslin and McDonald, 2004). *Acacia saligna* is at the same time one of the most planted non-timber woody species used for soil protection, reforestation, ornamental purposes, and for many other uses (Maslin and McDonald, 2004; Kull et al., 2011). In the Mediterranean, many *Acacia* species were introduced and planted mainly for stabilizing sand dunes and for preventing soil erosion (Del Vecchio et al., 2013). At present, *A saligna* is widespread in Mediterranean climates in its native range (Australia) and as an invasive non-native species (e.g., Algeria, Chile, Cyprus, Israel, Italy, Morocco, Portugal, South Africa and Spain, Thompson et al., 2015) as well as in other areas with seasonally dry conditions (e.g. Kenya) where it invades a great variety of habitat types (Le Maitre et al., 2000; Lorenzo et al., 2010a b; Boudiaf et al., 2013; Hernández et al., 2014; Lazzaro et al., 2014; Celesti-Grapow et al., 2016).

Invasion of *A. saligna* has detrimental effects on biodiversity and ecosystem functioning. *Acacia saligna* invaded areas are characterized by dense thickets (Lehrer et al., 2013) in which natural biodiversity is significantly modified (Del Vecchio et al., 2013). In addition to this, *A. saligna* invades several habitats of conservation value (Stanisci et al., 2012) and protected areas (Pinna et al., 2015; Acunto et al., 2017). Furthermore, it alters the runoff on slopes, modifies nutrient cycles and soil properties and decreases the aesthetic and recreational value of invaded landscapes (Brundu et al., 2019). For these reasons, in the European Union, *A. saligna* has been recently included in the list of invasive alien species of Union concern (Regulation (EU) No. 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species, hereafter, IAS Regulation). In addition to a ban on trade, planting and use *A. saligna*, member states are committed to surveillance to record occurrence in the environment and prevent its spread into or within the Union (Beninde et al., 2015).

A few iSDM studies on *A. saligna* have been done for Europe (e.g., Gutierres et al., 2011; Brundu et al., 2019) and for the Italian mainland (Marzialetti et al., 2019). However, further modelling studies are particularly urgent for the Mediterranean islands that are very well-known hotspots of biodiversity (Vilà et al., 2006b; Fenu et al., 2015; Peruzzi et al., 2015) highly threatened by invasive alien species (Hulme, 2004; Brundu, 2013; Malavasi et al., 2018). In addition, *A. saligna* is a neophyte in Sardinia (Galasso et al., 2018) and has probably not yet invaded all potentially suitable areas. Thus, identifying the unoccupied areas at risk of invasion provides crucial information for surveillance, management and prevention of impacts across the entire island. Therefore, this study aims to disentangle the role of propagule pressure, abiotic and biotic factors on the occurrence of *A. saligna* in the Mediterranean island of Sardinia. Our results provide an approach for prioritization of prevention, monitoring and control efforts towards areas more susceptible to be invaded, which would optimize the costs and time devoted to managing alien species.

2. Materials and methods

2.1. Study area

This study was conducted on Sardinia (Italy), the second largest island of the Mediterranean basin (24,100 km^2) (Fig. 1). The elevation ranges from 0 to 1,834 m a.s.l. (Punta la Marmora, Gennargentu massif). The climate is characterized by two main seasons, a hot-dry season and a cold-humid one. Annual mean temperature ranges from 17-18 °C on the coast to 10-12 °C on the inland mountains (Arrigoni, 2006). Annual precipitation varies greatly from the coast to the inland, from around 433 mm y^{-1} in the southern coast to 1,412 mm y^{-1} in the North at 1000 m a.s.l. (Arrigoni, 2006). In addition, a summer period of aridity, with low precipitation, marks the Sardinian climate typical Mediterranean *pluviseasonal-oceanic* (Rivas-Martinez and Rivas-Saenz, 1996-2019).

The coastal dunes of Sardinia harbor many ecosystems of priority conservation concern in Europe, listed by the "Habitats" Directive 92/43/EEC (e.g., HD 2250* - Coastal dunes with *Juniperus* spp., HD 2130* - Grey dunes, HD 2270*- Wooded dunes with *Pinus pinea* and/or *P. pinaster*). Importantly, the wooded dunes with *P. pinea* and/or *P. pinaster* in Italy and Sardinia are planted forests established for land reclamation and to protect agricultural areas and roads from sand (Falcucci et al., 2007; Malavasi et al., 2013). Besides invasive alien species, Sardinian and Mediterranean coastal ecosystems are jeopardized by a number of anthropogenic pressures (Falcucci et al., 2007; Malavasi et al., 2013) and widespread erosion (Drius et al., 2013; Camarda et al., 2015; Acosta et al., 2007; Malavasi et al., 2018).

2.2. Study species

Acacia saligna (Labill.) H.L.Wendl (Fabaceae) is an alien species that invades a large number of natural ecosystems in Sardinia and in the Mediterranean such as sand dune vegetation (e.g., Arrigoni, 2010; Gutierres et al., 2011; Meloni et al., 2013), and riparian plant communities (Lorenzo et al., 2010a; Del V ecchio et al., 2013; Lazzaro et al., 2014; Celesti-Grapow et al., 2016).

In Sardinia and in other regions in Italy, *A. saligna* was massively planted in the 1950's (Pavari and de Philippis, 1941; Del Vecchio et al., 2013) to stabilise sand dunes and protect *Pinus* spp. plantations from wind and sea spray (Maniero, 2000; Celesti-Grapow et al., 2009; Del Vecchio et al., 2013) and as an ornamental plant. In the invaded areas *A. saligna* forms dense thickets, including within wooded pine dunes (HD 2270*) and Mediterranean scrublands (HD 2260; Del Vecchio et al., 2013; Marzialetti et al., 2019). In addition, *A. saligna* outcompetes many Sardinian endemic species, in particular *Anchusa crispa* Viv. subsp. *maritima* (Vals.) Selvi & Bigazzi (Farris et al., 2013) typical of fixed coastal dunes with herbaceous vegetation (grey dunes - HD 2130*) and invades coastal dunes with *Juniperus* spp (HD 2250*) (Pinna et al., 2015; Acunto et al., 2017). Through nitrogen-fixation, *A. saligna* thickets promote the establishment of ruderal and nitrophilous species, simplifying and homogenising native plant communities (Caruso, 2012; Calabrese et al., 2017).

Under a Mediterranean climate, *A. saligna* can grow with mean annual temperature ranging from 11 to 23 °C and with annual precipitations from 240 to 1160 mm (Maslin and McDonald, 2004). Its persistence in the invaded sites is promoted by vegetative propagation (suckering) and by the establishment of a large persistent seed bank characterized by physical dormant seeds (Mehta, 2000; Strydom et al., 2012; Abd El-Gawad and El-Amier, 2015, Cohen et al., 2018).

2.3. Acacia saligna occurrence data

We used 432 georeferenced presence records of *A. saligna* collected around the invaded areas in Sardinia, and all these sites were periodically monitored (period 2000-2018), every two years, to check the persistence of the populations and their invasive status, i.e. whether only planted, casual or naturalised (Brundu et al., 2003; Camarda et al., 2016; Galasso et al., 2018). Field observations were georeferenced by means of a portable GPS (Garmin GPS 12 channels) and crosschecked on Google Earth imagery. For modelling these presences, 1000 pseudo-absence records were randomly generated across the entire study area excluding the areas occupied by the patches of *A. saligna*. Pseudo-absences were located at least 100 m apart from each other and the presence records were masked using a buffer with a radius of 150 m. The procedure was implemented in QGIS environment (3.2. “Bonn” version 2018).

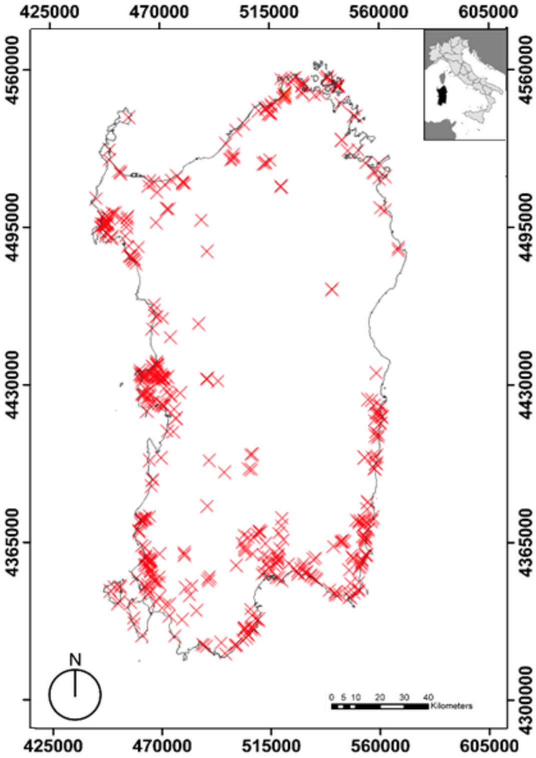

Figure 1. The study area and the distribution of the 432 *Acacia saligna* records (red crosses). The coordinate reference system is UTM (WGS84) zone 32 N.

2.4. Predictor variables

We selected a set of predictors for the presence of *A. saligna* acting as proxy variables for propagule pressure (P), abiotic (A) and biotic factors (B) (see Table 1 for detailed description).

The following variables have been used as measures of propagule pressure: (i) the locations in which *A. saligna* was planted for afforestation purposes in the past; (ii) the distance from roads (Le Maitre, 2004; Drake et al., 2015; Bazzichetto et al., 2016; Bazzichetto et al., 2018; Malavasi et al., 2018); and (iii) the extension of artificial surfaces (from CORINE land cover, CLC 2012). It is widely agreed that one of the main sources of propagule pressure for forest trees are tree nurseries and plantations (Malavasi et al., 2014), therefore, we classified the presence and pseudo-absence records as being inside or outside plantations of *A. saligna*. The locations of *A. saligna* afforestation were achieved from the published official maps of the Sardinian forest services (EFS, 2013), which at the moment are updated until 2013. We calculated the Euclidean distance of *A. saligna* wild populations from highways and primary roads due to the role of communication infrastructures in favoring alien species dispersal and the presence of planted individuals along the roads. Finally, in order to account for the dual role of urbanisations in providing new propagules from gardens, in which *A. saligna* is frequently planted (Carranza et al., 2010) and creating disturbed and bare areas more prone to invasion (Bazzichetto et al., 2018b) we considered the percentage of artificial surfaces (hereon ART; including urban fabrics; industrial and commercial units; mine, dump and construction sites, as defined in the Corine Land Cover CLC 2012 map for Italy) converted into a raster layer by a moving window of 11 x 11 pixels (see Marzialetti et al., 2019).

The following abiotic factors (A) were considered: (i) slope; (ii) average temperature; (iii) the distance from coastline, and (iv) frequency of wild fires. We included in the model a slope map in degrees extracted from a 10 x 10 m digital elevation model (resampled from 1 x 1 m Lidar data) because it is generally considered a good surrogate of water accumulation in the soil (MacMilland and Shary, 2009) affecting the suitability for *A. saligna* (Le Maitre, 2004; Gutierres et al., 2011). The thematic layer on the mean annual temperature for Sardinia for the period 1971-2000 was provided by the *Agenzia Regionale per la Protezione dell'Ambiente della Sardegna* (ARPAS - http://www.sardegnaambiente.it/arpas/). The sea-inland stress gradient that drives invasion on coastal areas (Carranza et al., 2011; Bazzichetto et al., 2016, 2018) was measured as the Euclidean distance to the nearest seashore. Then, as *A. saligna* successfully colonizes burned areas in the Mediterranean region (Bell et al., 1993) we included in the analysis a raster layer with the total number of wild fire events from 2005-2016.

Concerning biotic factors (B) facilitating or avoiding *A. saligna* invasion (Marzialetti et al., 2019) we considered the abundance of different natural and seminatural vegetation types. We calculated the percentage of cover of the following categories: coastal dunes with *Pinus* spp. plantations (AFF); dunes with woody vegetation, and degradation stages (WDH) and dune vegetation (DUN) using a moving window of 11 x 11 pixels (supplementary Table 1).

For all 18 variables, raster grid maps for the whole island of Sardinia were produced at 10 x 10 m resolution using the WGS84 datum and UTM 32N projection system (EPSG code: 32632) (supplementary Table 2) . However, to minimize collinearity only 10 were used in the final model (Table 1) selected according to Variance Inflation Factor (VIF).

Table 1. Predictor variables selected for building the iSDM, serving as proxies of propagule pressure (P), abiotic (A) and biotic (B) factors, along with their detailed description, the data source and the original scale. For a detailed explanation of the land cover types see supplementary materials, Table 2.

PAB factors	Predictor variables	Detailed description of the predictor variables	Source of the predictor variables

Propagule pressure (P)	Artificial areas (ART)	Percentage of artificial areas (CLC 2012 class 1) within a 100 m radius circular buffer	Corine Land Cover (CLC) 2012 vector map (scale 1:100000) (https://land.copernicus.eu/pan-european/corine-land-cover)
	A. saligna afforestation	Euclidean distance (m) from *Acacia* plantations	EFS (Ente Foreste della Sardegna 2013) (10 m spatial resolution)
	Road distance	Euclidean distance (m) from highways and primary roads	Regional geodatabase (scale 1:25000) (http://dati.regione.sardegna.it)
Abiotic (A)	Coastline distance	Euclidean distance (m) from the coastline	Regional geodatabase (scale 1:2000) (http://dati.regione.sardegna.it)
	Slope	Degrees, thematic layer produced by GIS analysis from a DEM with 10 x 10 m geometric resolution	Regional topography geodatabase (http://www.sardegnageoportale.it)
	Temperature	Annual mean temperature (°C)	Original raster layer produced by *Agenzia Regionale per la Protezione dell'Ambiente della Sardegna* - ARPAS (250 m spatial resolution)
	Fire frequency	Number of wildfire events in the period 2005-2016	Regional wildfire geodatabase (vector format) (scale 1:25000) (http://www.sardegnageoportale.it)
Biotic (B)	Afforestation(AFF)	Percentage of *Pinus* plantations (CLC 2012 class 3.12) within a 100 m radius circular buffer	Corine Land Cover (CLC) 2012 vector map (scale 1:100000) (https://land.copernicus.eu/pan-european/corine-land-cover)
	Dune vegetation (DUN)	Percentage of dune vegetation (CLC 2012 class 3.31) within a 100 m radius circular buffer	Corine Land Cover (CLC) 2012 vector map (scale 1:100000) (https://land.copernicus.eu/pan-european/corine-land-cover)
	Woody dune habitat (WDH)	Percentage of dunes with woody vegetation, and degradation stages (CLC 2012 class 3.2) within a 100 m radius circular buffer	Corine Land Cover (CLC) 2012 vector map (scale 1:100000) (https://land.copernicus.eu/pan-european/corine-land-cover)

2.5. Invasive alien species distribution model iSDM

We modeled the relationship between *A. saligna* occurrence and the PAB predictor variables (Table 1) using a Generalized Linear Model (GLM, dismo R package 1.1-4, Hijmans et al., 2017). We first extracted the PAB values at the presences and pseudo-absence records. We set the presence/pseudo-absence of the invasive alien plant as response variable and PAB predictors as covariates. Then we computed the Variance Inflation Factor (VIF, usdm R package, Babak, 2017) in order to exclude multi-collinearity between PAB proxy variables (Guisan and Thuiller, 2005). A predictor was excluded for VIF values higher than 3 (see supplementary Table 3 for collinearity analysis and variables selection). We fitted the GLM implementing when necessary polynomial transformations for non-linear responses (Venables and Ripley, 1994).

2.5.1. Model evaluation and predictions

We evaluated the performance of the model by the area under the receiver operator curve (AUC) (Pearce and Ferrier, 2000). AUC represents the probability that a randomly selected presence has a higher model-predicted suitability than a randomly selected background location (Manel et al., 2001). Specifically, for cross-validating the model we randomly partitioned the data and fitted the GLM 100 times, each time selecting 75% of points for model training and the remaining 25% for testing prediction accuracy. The iSDM predictive performance was summarized by averaging the cross-validated AUC values (LeDell et al., 2015). In addition, we obtained the goodness-of-fit of the model using the Nagelkerke R^2 (Nagelkerke,1991), which estimates the proportion of variance explained by the iSDM.

Finally, in order to schematically summarize the main trends and areas of invasibility in the island of Sardinia we projected the probabilities of invasion in the study area, and we classified them into five classes ranging from very low to very high (very low = suitability < 0.1, low = suitability ≥ 0.1 and < 0.3, intermediate = suitability ≥ 0.3 and < 0.5, high = suitability ≥ 0.5 and < 0.7, very high = suitability ≥ 0.7). Then for each *A. saligna* suitability class we calculated the respective percentage relative to the island extent.

3. Results

PAB predictors and Acacia saligna occurrence

The fitted GLM explained 75% of the variation in occurrence (Nagelkerke $R^2 = 0.75$) and had excellent predictive power (cross-validated mean AUC = 0.94 ± 0.007 sd). The model underlined the specific role of propagule pressure, abiotic and biotic factors in determining *A. saligna* occurrence across the island of Sardinia (Fig 2; Tab. 2).

Among propagule pressure (P) proxy variables, *A. saligna* tends to preferentially occur in areas with higher levels of urbanisation, close to roads and close to areas in which it has been planted (Fig. 2, Table 2). Among abiotic factors (A), *A. saligna* has a significant relationship with coastline distance and slope, indicating its preference for coastal and flat areas with moisture accumulation. *A. saligna* also preferred areas with low fire frequency and warmer conditions (Fig. 2, Table 2). All the biotic factors (B) showed a significant relationship with *A. saligna* occurrence. In relation to *Pinus* spp. plantation cover (AFF), *A. saligna* exhibited a parabolic trend with maximum suitability at around 40% AFF cover. *A. saligna* also prefers areas with woody vegetation (WDH) and semi-natural dune vegetation (DUN) (Fig. 2, Table 2).

The suitability map for Sardinia produced from the model (Fig. 3) yielded suitability classes for the whole island in the following proportions: 4.8 % very high, 4.7 % high, 5.8 % intermediate, 10.4 % low, and 74.3 % very low. Higher probabilities of *A. saligna* occurrence are located close to urban areas and roads, in coastal areas and on flat slopes (Fig. 3).

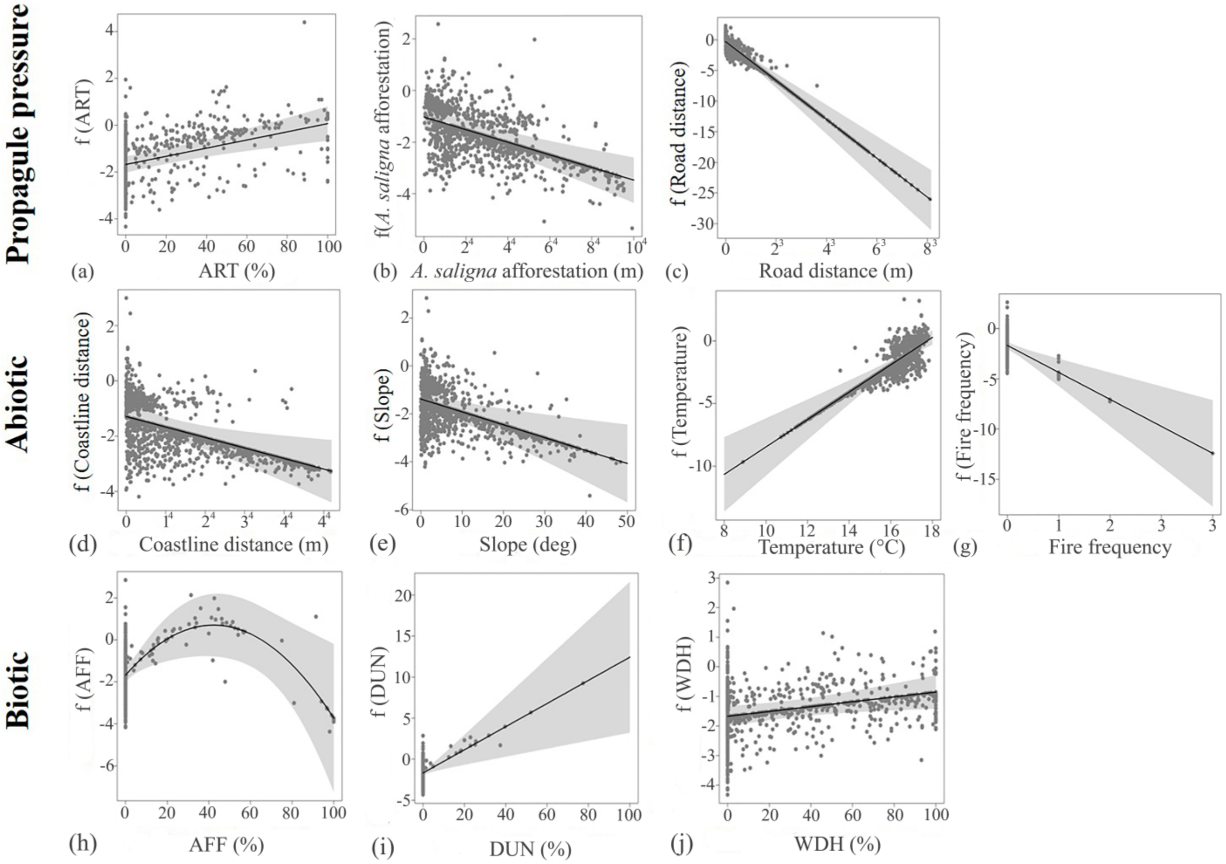

Figure 2. Regression plots along with confidence intervals (CI, grey shadowed area) showing the relationship between *Acacia saligna* occurrence (grey dots) and the PAB predictor variables: a) percentage of artificial areas, b) *A. saligna* afforestation distance, c) road distance, d) coastline distance, e) slope, f) annual mean temperature, g) fire frequency, h) percentage of *Pinus* sp. afforestation, i) percentage of herbaceous dune natural vegetation and j) percentage of dunes with woody vegetation, and degradation

stages (see Table 1). Predictor values are shown on the x-axes while partial suitability values are plotted on the y-axis (fitted value (f) of predictor variables).

Table 2. Results of the GLM analysis of *Acacia saligna* presence/pseudo-absence, modelled using predictors relating to propagule pressure (P), abiotic (A) and biotic (B) factors. Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’.

Predictors	**Estimate**	**Std. Error**	**Z value**	**p-value**
Intercept	-1.63E+01	2.95E+00	-5.519	3.41 10^{-8} ***
Propagule pressure (P)				
ART	1.74E+00	4.07E-01	4.288	1.80 10^{-5} ***
A. saligna afforestation	-3.07E-05	5.36E-06	-5.728	1.02 10^{-8} ***
Road distance	-3.29E-03	3.23E-04	-10.172	< 2 10^{-16} ***
Abiotic (A)				
Coastline distance	-3.77E-05	1.35E-05	-2.797	0.00515**
Slope	-8.67E-02	1.66E-02	-5.21	1.89 10^{-7} ***
Temperature	1.09E+00	1.76E-01	6.228	4.73 10^{-10} ***
Fire	-2.74E+00	6.83E-01	-4.014	5.96 10^{-5} ***
Biotic (B)				
AFF	-1.53E+01	5.36E+00	-2.858	0.00426**
DUN	1.41E+01	4.72E+00	2.99	0.00279**
WDH	8.21E-01	3.48E-01	2.359	0.0183*

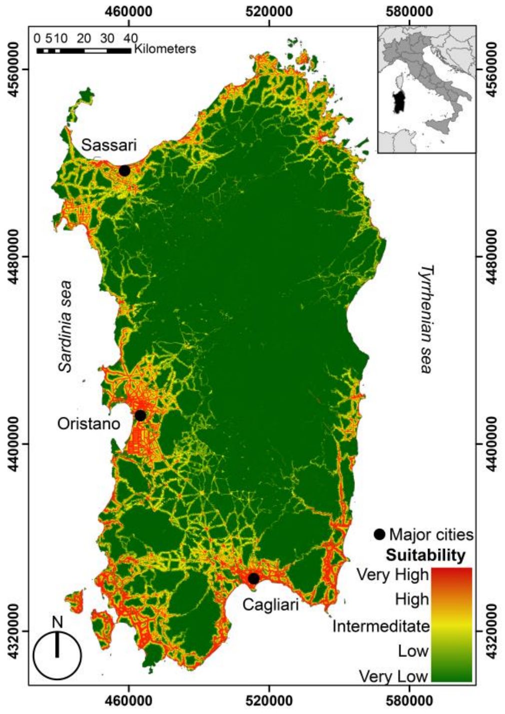

Figure 3. Suitability map for *Acacia saligna* in Sardinia based on the regional-scale invasive species distribution model (iSDM). Shading shows model predicted relative probabilities of occurrence. The coordinate reference system is UTM (WGS84) zone 32 N.

4. Discussion

This iSDM-based study explored whether 10 independent predictor variables explained the distribution of the invasive populations of *A. saligna* in the Mediterranean island of Sardinia and provided a high-resolution suitability map as a tool for managing this highly invasive species. Strong modelled responses to the predictor variables demonstrated the importance of propagule pressure (P), abiotic (A) and biotic (B) factors in determining suitability for *A. saligna* invasion. The predictive performance of the model, according to the cross-validated AUC, was very high (mean AUC = 0.94), allowing us to produce a highly informative high-resolution suitability map from the model.

Predictors relating to propagule pressure gave the strongest explanation of the occurrence of *A. saligna* in Sardinia. Specifically, our results show that proximity to *A. saligna* plantations, road networks and artificial or urbanised areas are important drivers of invasion. Indeed, we observed a higher chance of invasion close to past *A. saligna* plantations or roads and with medium coverage of built areas. The occurrence of *A. saligna* records close to the plantations established in the 1950s for stabilizing sand dunes and for other purposes (Pavari and De Philippis, 1941; Celesti-Grapow et al., 2009; Del Vecchio et al., 2013) suggests the species has spread from those plantations and confirms the importance of enforcing the ban on *A. saligna* introduction and further plantation enshrined in the IAS Regulation. The preference we found for road infrastructure and urbanised areas also suggests these locations increase the chance of propagule dispersal and subsequent establishment of *A. saligna*, similarly to patterns reported for many other IAS (e.g., Alpert, 2006; Hobbs et al., 2009, Wilson et al., 2011) including other *Acacia* spp. invasions in Europe (Gutierres et al., 2011; Marzialetti et al., 2019). Indeed, our results are consistent with previous results showing that invasive success of Australian acacias in general is correlated with propagule pressure and the extent of its use and dissemination within new regions (i.e. "human usage factors") (Castro-Díez et al. 2011).

Abiotic conditions were also very useful to explain the occurrence of *A. saligna* in Sardinia. The model shows that the invader thrives in warmer conditions with greater moisture accumulation and that are less fire prone (over the time period investigated). The model also found the species to be more common near to the coast. Overall, this is consistent with broader scale analysis showing that *A. saligna* has great potential to invade the Mediterranean area (Castro-Díez et al., 2011). However, previous findings showed that *A. saligna* avoids the highly stressful saline conditions found in immediate proximity to the seashore (Bazzichetto et al., 2016; Marzialetti et al., 2019) due to suppression of seed germination and seedling survival (Meloni et al., 2013). Concerning fire frequency during the investigated decade (2005-2016), we observed a higher suitability on non-burnt or burned only-once locations. Such apparent inconsistency with previous research that suggested an important role fires in explaining *A. saligna* distribution in the Mediterranean biome (Bell et al., 1993; Wilson et al., 2011) is probably related to the limited time interval for which mapped fires are available. The time series of fire occurrence might be not long enough to adequately describe the current invasion. Considering the observed preponderant role of propagule pressure and, that *Acacia saligna* afforestation dates back to the fifties, it is highly probable that also in Sardinia the fire have promoted invasions and favored germination from the seed bank (Richardson and Kluge, 2008) but in locations that have burnt before the analyzed decade (2005-2016). Our results suggest that, besides the utilization of high-resolution spatial data, the integration of temporal series data and landscape legacy could greatly help to further improve our knowledge on species invasions (Malavasi et al., 2014).

For Sardinia, biotic predictors also helped to explain the distribution of *A. saligna*, demonstrating preferences for sand dunes with open vegetation (DUN), sand dunes with woody vegetation (WDH) and plantations with intermediate cover of *Pinus* spp. (AFF). In these habitats *A. saligna* is competitive and has very clear negative impacts. As a result, management activities are in progress in these priority habitats, aiming for local eradication or population control, as defined by the European Directive 92/43/EEC, and for protection of critically endangered endemic species (IUCN 2001, 2003, 2006; Domina and Mazzola, 2008; Caruso, 2012; Del Vecchio et al., 2013; Brundu, 2013). Similar incidence of *A. saligna* on bare lands, and sparsely scattered vegetation has been described for arid ecosystems with sandy substrate in other regions (e.g., South-African fynbos, coastal sand dunes of Israel; Mehta, 2000; Bar Kutiel et al., 2004), and it could be explained by low competition with native species and efficient water uptake by *A. saligna* (Witkowski, 1991; Yelenik et al., 2004).

As well as providing understanding of the factors limiting *A. saligna* invasion, the model also allowed us to produce a high-resolution risk map for the whole island of Sardinia. The projected suitability map suggests a high risk of invasion in proximity to sand dunes, in the coastal plains and close to roads and other areas with strong human influence (see Angiolini et al., 2013). These results could help to optimize monitoring and prevention efforts, and to improve the existing management practice aimed at containing the invasion. For instance, we suggest directing early-warning monitoring campaigns along roads and railways as well cleaning and maintaining transportation infrastructure borders in order to reduce the presence of open disturbed areas in which seedlings can establish and spread. Wild fires or prescribed burning should also be limited as much as possible in any habitat where *A. saligna* is already established as occasional fires might strongly enhance seed germination from the soil seed-bank. In addition, we recommend the gradual removal of *A. saligna* from private and public gardens, botanic gardens or arboreta and other plantations from which they may escape and spread towards and establish within uninvaded habitats (Brundu et al., 2019).

The high and unrealized invasion risk also supports the recent inclusion of *A. saligna* in the list of invasive alien species of European Union concern, banning its intentional introduction in the European Union under article 7.1 of the IAS Regulation. However, the expected efficiency of these prevention measures may be of moderate effectiveness as *A. saligna* is already present in most of the EU Member States (Brundu et al., 2019). In fact, *A. saligna* could be declared a *widespread* species in several Member States (e.g., Cyprus, Croatia, France, Greece, Italy, Malta, Portugal and Spain). Under article 3 (point 16) of the IAS Regulation, a widespread species is an "*invasive alien species whose population has gone beyond the naturalization stage, in which a population is self-sustaining, and has spread to colonize a large part of the potential range where it can survive and reproduce*". For such widespread species it is very likely too late to apply eradication, except in restricted and priority areas. The majority of Member States shall have to put in place effective management (art. 19 of the IAS Regulation), so that their impact on biodiversity, the related ecosystem services, and, where applicable, on human health or the economy are minimized. Nevertheless, prohibition measures should limit further entry and introduction of new genotypes or provenances, and limit spread and re-invasion in sites where removal or control intervention are taking place. Finally, these prevention measures should be accompanied as much as possible by informative campaigns aiming to inform citizens, to increase public awareness, as unaware citizens frequently contribute to spread the invasive species (Brundu et al., 2019).

5. Conclusion

The iSDM developed based on high resolution thematic layers representing a range of PAB predictors explained the current distribution of *A. saligna* in Sardinia to a high degree of predictive accuracy. The model identified the important roles of propagule pressure, abiotic conditions and biotic factors in determining invasion risk and allowed the production of a suitability map for the Sardinian territory identifying locations at risk of further invasion. Such methodology could be further used for regional-scale modelling of other invasive species, including those listed in the IAS Regulation. We are convinced that our results and the chosen methodology match the demand of the Regulation for new early warning tools i.e. for predicting the location of new outbreaks, for establishing priorities for monitoring and control of widespread invasive species, and confirm the usefulness of predictive models for IAS management.

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