Risk Assessment:

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ACACIA SALIGNA

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Summary of the Express Pest Risk Assessment for *Acacia saligna*

PRA area: *European Union excluding outermost territories*

Main conclusions

The results of the PRA show that *A. saligna* poses a **high risk** to the endangered area within the European Union under current climate (i.e. significant parts of the Mediterranean Biogeographical region, but also countries along the Atlantic and the Black sea coasts for the '*pruinescens*' subspecies), with a low uncertainty (figure 5 in Appendix 4). Impacts in the current introduced range are high, and although the risk of further introduction in the European Union is considered as low, there is a moderate perceived risk of spread from established populations, facilitated by water and movements of soils contaminated by seeds or fragments of root suckers. Furthermore, the endangered area is likely to increase a lot during the coming decades due to climate change (figure 6 in Appendix 4).

Entry and establishment

A. saligna is already established in the endangered area within the European Union. It is a widespread IAS in the coastal areas of Cyprus, Italy, Portugal and Spain; it is also recorded from Croatia, France, Greece and Malta, but on a more sporadic basis. *A. saligna* is still absent from Bulgaria, Slovenia and Romania, although appropriate climatic conditions and habitats are encountered. The risk of further entry into the region as seeds and plant for planting is considered low with a low uncertainty. The potential for establishment in both the natural and managed environment is high with a low uncertainty. This potential is known to be favoured by fire and soil disturbance that create suitable conditions for germination (breaking seed dormancy) and establishment of seedlings of *A. saligna*.

Potential impacts in the PRA area

Impacts on biodiversity are likely to be similar in the PRA area as to those documented in the current area of distribution (high with a low uncertainty). In Cyprus, Italy, Malta and Portugal, *A. saligna* forms extensive dense stands which can exclude most native plant species and change community composition, especially in coastal sand dune and riparian ecosystems. Impacts on several Red Data Book species in the EU are expected such as for *Aegilops bicornis, Anchusa crispa* subsp. *maritima* and *Anthyllis hermanniae* subsp. *brutia*.

Impacts on ecosystem services will be similar to those seen in the current area of distribution (high with a moderate uncertainty). *A. saligna* persistently transforms ecosystems and their disturbance regime through reinforcing feedback processes. It affects provisioning (reduction of surface runoff and soil water reserves), regulating and supporting (modification of nutrient cycling and soil properties) and cultural services (reduction of aesthetic and recreational landscape quality). It may also increase fire intensity and frequency under extreme climatic conditions.

Socio-economic impacts will be similar in the PRA area as to those seen in the current area of distribution (high with moderate uncertainty), due e.g. to the very high costs caused by a strong hydrological impact (loss of water provision) and its long-term management.

Climate change

Climate change scenario RCP8.5 is predicted to increase suitability dramatically and to cause a strong expansion of the endangered area within the European Union. Major parts of the Mediterranean, Black Sea, Atlantic and Continental biogeographical regions will be at risk for all the different subspecies; it is also predicted that the '*lindleyi*' and the '*pruinescens*' subspecies will be able to establish in a wider range, including a larger part of the Continental biogeographical region and most of the Pannonian biogeographical region (see figure 6 in Appendix 4). Climate change is also expected to alter the geographic distribution of wildfire, a process that could promote further establishment of *Acacia saligna* close to plantations and invaded sites.

Socio-economic benefits

While the plant is traded as an ornamental, as forestry species or for other uses including honey production, the value it currently generates within the European Union is limited and benefits it produces are unlikely to exceed the cost of negative impacts it causes. Moreover, alternative species are available. Future profits generated by biomass production on marginal soils are expected to be limited due to suboptimal growth conditions and accompanied by high profitability uncertainty.

Phytosanitary risk for the *endangered area*: HIGH

Level of uncertainty of assessment: LOW

Other recommendations:

With the exception of South Africa, very limited efforts have been conducted in the invaded range and in the European Union to distinguish among the different subspecies or variants described for *Acacia saligna*. Other Australian acacia species (e.g. *A. dealbata, A. longifolia, A. mearnsii* and *A. melanoxylon)* are introduced and planted for various purposes within the European Union and some of them are reported to colonise natural environments. An accurate assessment of their invasiveness should be conducted before further use.

Express Pest Risk Assessment

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Stage 1. Initiation

1.1 - Reason for performing the Pest Risk Assessment (PRA)

Acacia saligna (Labill.) H.L.Wendl *s.l.*[1], (Coojong wattle) is considered the most widely planted non-timber woody species for multiple purposes including afforestation/reforestation, ornamental use and soil protection, but also for fuelwood, charcoal, fodder, tannin and biomass production and other uses (Maslin and McDonald, 2004; Griffin *et al.*, 2011; Kull *et al.*, 2011). This evergreen species covers an estimated 600,000 hectares worldwide and has been widely cultivated within and outside its native range also in Australia (Maslin and McDonald, 2004; Griffin *et al.*, 2011). However, it is considered an invasive alien species in several regions in the world characterized by Mediterranean-type climate, such as parts of Australia, Algeria, Chile, Cyprus, Israel, Italy, Kenya, Morocco, Portugal, South Africa and Spain where it causes strong and persistent impacts on biodiversity and ecosystem services (e.g., Thompson *et al.*, 2015). Similarly, within the European Union, *A. saligna* has been introduced in a significant number of Member States. It is often considered invasive and many LIFE projects are actively promoting local eradication and control of *A. saligna* in protected areas to restore native plant communities or endemic and endangered native species. Therefore, the present PRA aimed to collect and analyse information on the invasive risk of further introduction and spread of *A. saligna* in the PRA area, *i.e.* in the European Union as defined in the framework of the Regulation (EU) No. 1143/2014[2].

1.2 - PRA area

The PRA area being assessed is the European Union, as defined in the framework of the Regulation (EU) No. 1143/2014.

1.3 - PRA scheme

This Express Pest risk assessment document follows EPPO Standard PM 5/5(1) ***Decision-Support Scheme for an Express Pest Risk Analysis***, with modification and integrations for section 12 and section 15, to take into account the criteria for risk assessment required by the Reg. (EU) No. 1143/2014 (see Roy *et al.* 2014, Invasive alien species – framework for the identification of invasive alien species of EU concern. ENV.B.2/ETU/2013/0026 and Roy *et al.*, 2017). This amended scheme has been utilised during the LIFE project IAP-RISK (http://www.iap-risk.eu/) on sixteen alien plants; it is not yet an EPPO standard, but it is under consideration to be formally approved as such. The authors of this PRA consider this scheme as reliably suitable to fulfil all the requirements of the Reg. (EU) No. 1143/2014. The biogeographical regions are herewith considered according to the official delineations used in the Habitats Directive (92/43/EEC) and for the EMERALD Network set up under the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention).

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[1] (*s.l.* = sensu lato - in the broad sense), *Cf.* sections 2.1.1 – 2.1.5 for details.
[2] 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.

Stage 2. Pest risk assessment

2.1 - Taxonomy and identification

2.1.1 - Taxonomy

Kingdom	*Plantae*
Subkingdom	*Tracheobionta* (Vascular plants)
Superdivision	*Spermatophyta* (Seed plants)
Division	*Magnoliophyta* (Flowering plants)
Class	*Eudicotyledons*
Subclass	*Fabids*
Order	*Fabales* Bromhead, Edinburgh New Philos. J. 25: 126. (1838)
Family	*Fabaceae* Lindl., Intr.Nat.Syst.Bot. Ed. 2: 148 (1836), *nom. cons.* = *Leguminosae* Juss., *nom. cons* *Leguminosae*, LPWG (2017)
Subfamily	*Caesalpinoideae – Acacia* clade, LPWG (2017)
Genus	*Acacia* Mill. *s.l*, Gard. Dict. Abr. ed. 4 (1754), *nom. et typ. cons*.

Acacia saligna (Labill.) H.L.Wendl., Comm. Acac. Aphyll. 26. 1820 (Family *Leguminosae*, LPWG, 2017) is a native (endemic) Western Australian very polymorphic species (Maslin, 1974) with a widespread but naturally patchy distribution currently circumscribed by four to five informal subspecies (Millar *et al.*, 2010; WorldWideWattle ver. 2, 2017). The accepted name is based on *Mimosa saligna* Labill., Nov. Holl. Pl. 2: 86, t. 235. 1806 (basionym). The lectotype for the name was selected by B.R. Maslin (1974) among the samples collected by Labillardiere and stored at the herbarium of Florence, Italy (FI). The specimen selected as lectotype represents the taxon later described as *Acacia cyanophylla* Lindl. (Edwards's Botanical Register 25 1839 Misc. 45, Misc. 45, No. 64) which is therefore a taxonomic synonym (homotypic synonym) of *A. saligna*.

As a result of its polymorphism, four genetic lineages or subspecies have been described, consistent with the morphological groupings of the species complex: *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***saligna*** (autonym), *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***stolonifera*** M.W.McDonald & Maslin ms, *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***pruinescens*** M.W.McDonald & Maslin ms [and *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***lindleyi*** (Meisn.) M.W.McDonald & Maslin ms (Maslin *et al.*, 2006; https://florabase.dpaw.wa.gov.au/). These four subspecies can be distinguished by a combination of morphological differences including phyllode appearance, the shape of the inflorescence bud, the length of racemes and the diameter, colour and number of flower heads (M. McDonald personal communication, in Millar *et al*. 2011). According to this morphological grouping of the species complex, each subspecies is geographically associated with a particular ecological habitat as described in the pest overview section (Section 2.2) (Thompson *et al.*, 2011, 2015). The taxonomy and nomenclature of *Acacia saligna* s.l. is under ongoing revision in Australia. At the same time, the concept of ‘variant’ is found in the scientific literature and in technical reports, or in provenance trials. Importantly, (1) subsp. ***lindleyi*** is also referred to as the 'typical' variant; (2) subsp. ***pruinescens*** is referred to as the 'Tweed River' variant; (3) subsp. ***saligna*** is referred to as the 'cyanophylla' variant and (4) subsp. ***stolonifera*** is referred to as the 'forest' variant (Maslin *et al.* 2011) (Table 2, Section 2.2.2).

Genetic divergence is evident between these subspecies (Millar *et al.*, 2012 and references cited therein), which encompass a wide range of morphological variation and show a high degree of morphological plasticity. Natural hybridization is uncommon in Australia due to the disjunct distribution of populations and limited areas of natural sympatry of the subspecies but has been confirmed in mixed plantations using molecular markers (Millar *et al.*, 2012 and references cited therein). The *A. saligna* subspecies can be distinguished by a combination of morphological differences including phyllode appearance, the shape of the inflorescence bud, the length of racemes and the diameter, colour and number of flower heads (Millar *et al.*, 2008b and references cited therein); however, these characteristics can only be assessed when plants are suitably mature and only while plants are developing buds or flowering (Millar *et al.*, 2008b and references cited therein). The subspecies of *A. saligna* display variation in key traits, such as seed set, fecundity and suckering (Millar *et al.*, 2008b and references cited therein) that are all important aspects to consider both for the identification and for assessing the invasion risk and the phytosanitary measures.

These four informal subspecies were recently and tentatively reclassified into three major subspecies lineages: subsp. *lindleyi*, 'subsp. *pruinescens* + subsp. *saligna*' and subsp. *stolonifera* (Maslin *et al.*, 2011; Millar *et al.*, 2011;). However, according to the inflorescence characters Maslin *et al.* (2011), have proposed also only two-groups ('subsp. *pruinescens* + subsp. *saligna*' and 'subsp. *lindleyi* + subsp. *stolonifera*'). As a result, the identification of *A. saligna* subspecies is challenging (Le Houerou and Pontanier, 1987; Maslin and McDonald, 2004; Millar *et al.*, 2008b; Millar *et al.*, 2011).

Finally, *Acacia provincialis* was described from cultivated material and was said by its original authors to represent a hybrid between *A. retinodes* and *A. cyanophylla* (= *A. saligna*); having inspected these original specimens Maslin & McDonald (2004) state that they appear to be *A. retinodes* 'swamp' variant; these authors - in fact - consider very unlikely that hybrids between *A. retinodes* and *A. saligna* would naturally occur.

2.1.2 - Main synonyms

The main synonyms have been retrieved from the web site "The Plant List[3]", as follows:

Mimosa saligna Labill., Nov. Holl. Pl. 2: 86, t. 235 (1807) (basionym);

Acacia bracteata Maiden & Blakeley, Roy. Soc. W. Australia 13: 18, t. 10, figs 7–11 (1928);

Acacia cyanophylla Lindl., Edward's Bot. Reg. 25: Misc. 45 (1839);

Acacia lindleyi Meissner, in J.G.C.Lehmann, Pl. Preiss. 1: 14 (1844);

Racosperma salignum (Labill.) Pedley, Austrobaileya 2: 355 (1987).

2.1.3 - Common names

Coojong wattle, golden-wreath wattle, orange wattle, blue-leafed wattle, Port Jackson willow; Acacia azul (Spanish) Akacja (Maltese); Acacia saligna (Italian); Mimosa bleuâtre (French).

2.1.4 - Main related or look-alike species

A. saligna has no known close relatives in the European Union, but it resembles, superficially, a number of other introduced *Acacia* species including *A. pycnantha* (Maslin, 1974), however the latter is distinguished by its stouter raceme axes and peduncles, its prominently tapered phyllode bases, it smaller pulvinus, and its smaller glands. In its growth habit, phyllode morphology, glabrous raceme, and large

[3] http://www.theplantlist.org/tpl1.1/record/ild-591 [Accessed 15 December 2017].

flower heads, *A. saligna* superficially resembles *A. amplices* B.R.Maslin; however, the flowers, legumes, and seeds of these two species are quite different. Finally, *A. saligna* can be occasionally confused with *A. microbotrya* Benth. and *A. rostellifera* Benth. (Maslin, 1974). It might also be superficially confused with *Acacia retinodes* Schltdl. Importantly, *A. pycnantha* (native to Australia) is considered invasive in many Mediterranean countries, including Italy (e.g., Giovanetti et al., 2015) thus it should not be considered as a substitute species.

2.1.5 - Terminology used in the present PRA for taxa names

In the present PRA the terms "*Acacia saligna"* and/or "*Acacia saligna* s.l." (s.l. = *sensu lato* - in the broad sense) (also abbreviated as *A. saligna*) both indicate the species complex, i.e. the whole group of subspecies (or lower taxa, such as, e.g. cultivated varieties, cultigens and provenances) that have been described for the entity *Acacia saligna* (Labill.) H.L.Wendl., Comm. Acac. Aphyll. 26. 1820[4]. Whenever the PRA refers to a subspecific entity (*cf*. section 2.1.1), its full name is reported. The present PRA addresses the risk posed by *Acacia saligna* s.l.

2.1.6 - Identification (brief description)

The following description has been retrieved from the web site "Flora of Australia On Line"[5].

Evergreen bushy shrub or tree mostly 2–6 (10) m high. Bark grey. Branchlets often pendulous, normally slightly flexuose, often pruinose (especially when young), glabrous. Phyllodes often pendulous, variable in shape and size, linear to lanceolate, straight to falcate, 7–25 cm long, (2–) 4–20 mm wide, often larger towards base of plant, green to glaucous, glabrous, with prominent midrib, finely penninerved (absent on very narrow phyllodes); gland ±disciform, 1–2 mm wide, 0–3 mm above pulvinus; pulvinus mostly 1–2 mm long, coarsely wrinkled. Inflorescences mostly 2–10-headed racemes, enclosed when young by imbricate bracts, with bract scars evident at anthesis; raceme axes mostly 3–30 mm long, glabrous; peduncles 5–15 mm long, glabrous; heads globular, mostly 7–10 mm diam. at anthesis and 25–55-flowered, golden. Flowers 5-merous; sepals c. 4/5-united. Pods linear, flat, shallowly constricted between seeds, 8–12 cm long, 4–6 mm wide, thinly coriaceous, glabrous. Seeds longitudinal, oblong to slightly elliptic, 5–6 mm long, shiny, dark brown to black; aril clavate.

2.2 - Pest overview

2.2.1 - Introduction

Acacia saligna is an evergreen shrub or small tree which grows to a height of 2-6 (10) m (Maslin, 1974; Degen *et al*., 1995; Virtue and Melland, 2003), native and endemic to Western Australia. It is a fast-growing species characterized by both clonal propagation and sexual reproduction; it is well adapted to semiarid environments and is fire-resilient. *A. saligna* has a mixed mating system, preferential outcrossing, but also with a certain level of selfing (George *et al.*, 2008). Under cultivation, it tends to have a short lifespan: typically, less than 10 years and in some instances less than 5 years in Australia (World Wide Wattle 2017[6]). However, an average lifespan of 30-40 years has been reported for South Africa (Milton and Hall, 1981 as reported in Wood and Morris, 2007) The age of the flowering is two-three years. *A. saligna* has bright and dense yellow, globular flowerheads with a generalist floral morphology. Flowers are visited most frequently by bees, wasps, flies and beetles (Gibson *et al*., 2013). Actually, the fundamental floral morphology shared by all Australian acacias identifies a generalist entomophilous pollination syndrome as it provides accessible floral rewards to almost any insect visitor (Gibson *et al.*, 2011).

[4] *Acacia saligna* was described by Wendland, Heinrich Ludwig, in 1820 in "*Commentatio de Acaciis Aphyllis. Hannoverae*", vol. 4, pp. 26-27.
[5] http://www.anbg.gov.au/abrs/online-resources/flora/redirect.jsp
[6] http://worldwidewattle.com/infogallery/projects/saligna.php [Accessed 19 December 2017].

A. saligna s.l. flowers from (August) September to October (November) in the native range (Henderson, 2001; Australia Florabank 2017[7]). Flowering periods in the invaded range are reported in the following table:

Table 1: Flowering periods reported from the invaded range of *Acacia saligna*.

Location	Flowering period	Source
Chile (alien range)	July - October	Perret *et al.* (2001)
Italy, Sicily (alien range)	March - May	http://www.dipbot.unict.it/orto-botanico/scheda.aspx?i=356
Spain (alien range)	March - May	Flora Iberica – Paiva (1999)
South Africa (alien range)	August - September	Milton and Moll (1981)

Field observations in Portugal reported more hermaphrodite and male flowers which are easily identified by the presence or absence of a well-developed pistil. *A. saligna* showed lower investment in flower head production (despite the higher number of flowers per flower head) and the fecundity of all ovules in a flower is rare (e.g. mostly had only one seed per pod) (Correia *et al.*, 2014).

The maximum recorded value of annual seed rain of *Acacia saligna* in the invaded range (South Africa) is 5,443 seeds/m^2 (Milton and Hall, 1981 as reported by Richardson and Kluge, 2008). The vast majority of the seeds are added to the seed bank where they remain dormant until the testa is damaged or weathered sufficiently to be permeable to water and germinate (Milton and Hall, 1981). As a result, the maximum recorded value of seed bank of *A. saligna* in South Africa is 46,000 seeds/m^2 (Holmes *et al.*, 1987 as reported by Richardson and Kluge, 2008). In **Cyprus**, as reported in the final Report of the project LIFE12 NAT/CY/000758[8], several samples (25 x 25 cm) were taken from soil in three layers. The average number of seeds per square meter at the soil surface was estimated to be 1,648 seeds, at 0-10 cm depth was 2,160 seeds and at 10-20 cm was 400 seeds.

As for many *Acacia* species, seed biology syndromes are largely shaped by fire driven ecosystems that are present throughout much of Australia and the introduced range (Mediterranean-type climate regions). Fire-adaptive traits include: production of large quantities of hard-coated, heat-tolerant and long-lived seeds with the capacity for long dormancy in the soil (even for decades); stimulation of germination by heat and/or smoke; seed dispersal and burial by ants (Holmes, 1989, 1990b; Richardson and Kluge, 2008; Le Maitre *et al*., 2011; Dufour-Dror, 2012).

Fire is a key part of the life cycle of *A. saligna.* Fire stimulates seed germination in several invasive acacias such as *A. melanoxylon*, *A. dealbata* and *A. saligna* (García *et al.*, 2007; Lorenzo *et al.*, 2010a; Wilson *et al.*, 2011). On the contrary, the plant itself is absolutely fire sensitive, although resilient thanks to vegetative resprouts.

2.2.2 - Habitat and environmental requirements

In the native range *Acacia saligna* s.l. is widespread and often locally abundant and occurs principally in dry sclerophyll forest or temperate woodlands (Hall and Turnbull, 1976). In south-east Australia, *A. saligna* s.l. has established in coastal scrublands, grassy woodlands, heathlands, warmer moist forests and riparian areas (Muyt, 2001). However, according to the morphological groupings of the species complex

[7] http://www.florabank.org.au/lucid/key/species%20navigator/media/html/Acacia_saligna.htm [Accessed 22 December 2017].

[8] Final Report Covering the project activities from 01/09/2013 to 28/02/2017, Reporting Date, 28/02/2017, LIFE-RIZOELIA: Improving the conservation status of the priority habitat types *1520 and *5220 at the Rizoelia National Forest Park (http://www.life-rizoelia.eu/).

(see table 2), each subspecies is geographically associated with a particular habitat type: *A. saligna* subsp. *lindleyi* (watercourses, sand dunes, coastal plains), subsp. *pruinescens* (deep soil in swamp-like areas), *A. saligna* subsp. *saligna* (coastal plains) and *A. saligna* subsp. *stolonifera* (watercourses and forest-like areas) (Thompson *et al.*, 2011).

Table 2. An assessment of traits considered important from a domestication perspective for the *Acacia saligna* variants, based on observations from natural populations in native range (McDonald *et al.*, 2007).

	A. saligna* subsp. *lindleyi	***A. saligna* subsp. *pruinescens***	***A. saligna* subsp. *saligna***	***A. saligna* subsp. *stolonifera***
Variants	'Typical'	'Tweed River'	'Cyanophylla'	'Forest'
Size	Low-tall	Low-tall	Tall	Low-tall
Biomass production	Poor-good	Fair-good	Excellent	Poor-good
Coppicing ability	Poor-good	Fair	Excellent	Fair
Suckering ability	Weak-moderate	Strong	Weak-moderate	Strong
Lowest minimum t°	0 °C	-5 °C	-4 °C	-4 °C

As noted by Doran and Turnbull (1997) and Hobbs *et al.* (2009), *A. saligna* s.l. occurs on many soil types, especially deep poor and calcareous sands, but also on moderately heavy clays. In its natural habitat, the species is normally found near water courses and other wet areas. It mainly grows on coastal sand plains but extends to a wide variety of situations from swampy sites and river banks to small or rocky hills (often granitic) (Groves, 1994). Simmons (1981) reported that *A. saligna* tolerates alkaline and saline soils and a grows under a wide range of soil water regimes. However, its ability to fix nitrogen and its growth performances are greatly reduced by drought (< 350 mm annual precipitation), water-logging and shading (Nakos, 1977; NAS, 1980a; Maslin and McDonald, 2004; Hobbs *et al.*, 2009).

In its natural range within south-western Australia, *A. saligna* grows under a Mediterranean climate type, with a mean annual temperature range between 11 and 23 °C, minimum temperature range between 2 and 10 °C and maximum temperature range between 25 and 35 °C. The long-term average rainfall is 580 mm, with a range of 240 to 1160 mm, falling mostly in the winter months (Maslin and McDonald, 2004; Hobbs *et al.*, 2009).

In its introduced range, *A. saligna* is reported as established (i.e., naturalised[9]) in many semi-natural habitats within Mediterranean-type regions all over the world, such as riparian habitats, shrublands, fynbos (South Africa), forests, grasslands and sand dunes (Le Maitre *et al.*, 2000; Hadjikyriakou and Hadjisterkotis, 2002; Lorenzo *et al.*, 2010a; Del Vecchio *et al.*, 2013; Hernández *et al.*, 2014; Lazzaro *et al.*, 2014; Celesti-Grapow *et al.*, 2016: Souza-Alonso *et al.*, 2017).

Soil and climatic preferences observed in the introduced range are close to those described from the native range (Hobbs *et al.*, 2009; Thompson *et al.*, 2011). It has however been often planted in more arid conditions that those encountered in its native range, as it is the case in North Africa. In those conditions, *A. saligna* is reported to have a lower capacity to sucker and make dense thickets; its invasiveness and competitiveness are reduced by suboptimal growth conditions and possibly also absence of fire

[9] Naturalised = capable of establishing a viable population and spreading in the environment under current conditions and in foreseeable climate change conditions at least in one biogeographical region shared by more than two Member States (*sensu* Art. 4.3.b., Reg. EU No. 1143/2014).

perturbation (Tiedeman and Johnson, 1992; Le Houerou, 2000; Derbel *et al.*, 2009; Amrani *et al.*, 2010; Reubens *et al.*, 2011; Wilson *et al.*, 2011).

2.2.3 Resource acquisition mechanisms

A. saligna is especially competitive because of faster root and shoot growth amongst the group of Australian acacia species (Witkowski, 1994; Atkin *et al.*, 1998). In South African fynbos and in Australian drylands, it was shown to grow taller and faster than native vegetation due to very efficient resource acquisition mechanisms. It develops horizontal roots up to 12 m long as well as vertical roots that reach depths of 3.5 m, and up to 16 m in sandy habitats; its roots penetrate earlier and deeper in the soil profile than most other plants (Witkowski, 1991a; Musil, 1993; Dufour-Dror, 2012; Knight *et al.*, 2002). It also has efficient mycorrhizal and N_2-fixing symbioses that allows it to easily colonise nutrient poor soils (Hoffman and Mitchell, 1986; Musil, 1993; Stock *et al.*, 1995). Furthermore, sclerophylly and plant ability to remobilize limiting nutrients enable efficient nutrient conservation (Witkowski, 1991b; Morris *et al.*, 2011).

Field observations and laboratory experiments suggest that *A. saligna* also releases persistent allelopathic compounds in the soil from fallen leaves and flowers, plant leachates or root exudates (e.g. low vegetation cover and strong decrease of *Artemisia monosperma* plants in the vicinity of the tree) as also observed for other acacia species (El-Bana 2008, Abd El-Gawad and El-Amier, 2015).

2.2.4 - Symptoms

One of the primary symptoms of *A. saligna* in the non-native ranges is the tendency to make dense and persistent thickets and to cause a reduction in the species richness, native species cover, and changes in community structure (e.g., Holmes and Cowling, 1997; Richardson *et al.*, 1989). In many cases, the formation of dense stands occurs close to existing plantations with *A. saligna*, or can be the result of wildfires (Musil, 1993; Holmes and Cowling, 1997) or even prescribed fires. *A. saligna* not only outcompetes indigenous plant species by growing faster and taller, but it also transforms the environment by creating shady canopy cover and by altering soil properties through a combination of fixing nitrogen and its high input of leaf litter (Witkowski 1991; Holmes and Cowling, 1997). Dense litter layers under acacias also prevent native seed contact with the soil (Appendix 1, Figure 7). With a smaller proportion of seeds in the seed bank, many native species might regenerate poorly after a fire in comparison to *A. saligna*.

2.2.5 - Existing PRAs

Australia: Melland and Virtue (2002) applied the Animal and Plant Control Commission (APCC) Weed Assessment Scoresheet (Virtue, 2000) was used to rank the potential weed threats of *A. saligna* to native vegetation in the seven regions of South Australia. Scoresheet consists of a series of multiple choice questions, grouped into three criteria; Invasiveness, Impacts and Potential Distribution. Scores for the criteria (each ranging from 0 to 10) are then multiplied to give a Weed Importance score. On a state-wide scale, *A. saligna* scored a very high weed risk to native vegetation. More precisely, *A. saligna* poses a very high weed risk in the Eyre, Northern Agricultural Districts, Mt. Lofty Ranges/Metro and South East regions. The species poses a high weed risk in the Murray Darling Basin, and a negligible risk in the other regions, due to poor climate matches. In addition, *A. saligna* features among the most invasive garden plants in each state, territory and the whole of Australia that were available for sale in NSW in 2006 according to Coutts-Smith and Downey (2006). In Australia, 43 native acacias are naturalised beyond their native range (Adair, 2008).

France: Using the risk assessment system developed by Weber and Gut (2004) for central Europe (W-G - WRA), *A. saligna* has been identified as priority for a national PRA. *A. saligna* scored 31 out of 39 highlighting a high risk to the Mediterranean biogeographical region of France (Fried, 2010).

Hawaii: Pacific Island Ecosystems at Risk (PIER)[10]. This risk assessment predicts the likelihood of invasions of species in Hawaii, and the high islands of the Pacific. The risk assessment for Hawaii scored *A. saligna* as 17, indicating that the species poses a high risk of invasion.

Italy: Crosti *et al.* (2010) used a modified version of the Australian Weed Risk Assessment (A-WRA) adapted for the Mediterranean region of Central Italy, to assess the risk for a number of invasive alien plants in Lazio (Italy, Mediterranean biogeographical region). *A. saligna* scored 12, resulting in a "reject" decision according to the A-WRA.

Spain: Gassó *et al.* (2010) applied the Australian Weed Risk Assessment scheme (A-WRA) of Pheloung *et al.* (1999), modified for Spain, to 100 invasive and 97 casual[11] species in Spain. *A. saligna* scored 22, indicating a high risk and rejecting its import.

Socio-economic benefits

Introduction and use of *A. saligna* within the **European Union** mostly occurred in the past for reafforestation, firewood production, erosion control, soil stabilisation and protection purposes, especially in coastal dune ecosystems in the Mediterranean region and islands (Hadjikyriakou and Hadjisterkoti, 2002; Celesti-Grapow *et al.*, 2010; Marchante and Marchante, 2014), honey production and other secondary uses. Since recent years, its introduction for biomass production (short rotation coppicing systems) in marginal soil conditions under Mediterranean climates is under investigation in the European Union (Crosti *et al.*, 2010; Facciotto and Nervo, 2011) as in the rest of the world (Goslin and McDonald, 2006; Hobbs *et al*., 2009; Griffin *et al.*, 2011).

So far, few studies have specifically quantified both the resprouting capacity and the impact of nutrient and water availability on the biomass yields of the different subspecies of *A. saligna* (Maslin and Mc Donald, 2004; Hobbs *et al*., 2011). However, it is known that their growth rates and biomass production can vary markedly between and even within sites (Hobbs *et al.*, 2011). Field trials conducted in Chile (Perret *et al*., 2001), in Israel (Zegada-Lizarazu *et al*., 2007) and in Italy (Faccciotto and Nervo, 2011) suggest that water is an important limiting factor to the growth of *A. saligna* and that irrigation and potentially also fertilization will have to be applied to guarantee a high sustained yield in short rotation coppicing systems under Mediterranean climates. As in the cases of *Jatropha curcas*, *Robinia pseudoacacia* and other energy woody crops (Gasol *et al.*, 2010; Dauber *et al.*, 2012; Blanco-Canqui, 2016), it may be expected that *A. saligna* may not provide substantial economic benefits as a bioenergy crop due to limited growth and high installation costs in these conditions.

Similarly, *A. saligna* was widely planted for drift sand control and tannin production following its introduction to South Africa's Cape Floristic Region (CFR) in the 19th century. Mayer (1995) reports that the massive introduction of *A. saligna* took place in sand dune areas under the direction of the local Forestry Administration, with the initial aim of stopping the sand from moving. However, it has been also observed that Australian acacias often fail to adequately prevent soil erosion in several regions because of topsoil loss when harvesting as a consequence of absence of herbaceous vegetation beneath them; plantations for dune stabilisation may also destabilise the coastline and trigger massive beach erosion (Lubke, 1985; Carruters *et al.*, 2011; Low, 2012). In South Australia, it is also planted with other deep-rooted perennial plant species to reverse or control salinity in dryland habitats (Bennett and Virtue, 2005, Hobbs *et al*., 2009).

More in general, *Acacia saligna* has a long history of **multi-purpose use** in Australia and overseas. Of the 25 most exported Australian acacias, this medium-sized tree is the most widely planted non-timber species covering 600,000 ha worldwide (Griffin *et al.*, 2011; Thompson *et al.*, 2015). Under cultivation this species is capable of developing into a robust woody shrub or small tree, growing on a wide range of soils and producing a large quantity of woody biomass, foliage, (green) pods and seeds. Since the past it has been used for soil protection and desalination, mine site rehabilitation, revegetation, agroforestry,

[10] http://www.hear.org/pier/wra/pacific/Acacia_saligna.pdf

[11] Casual = Alien plants that may flourish and even reproduce occasionally in an area, but which do not form self-replacing populations, and which rely on repeated introductions for their persistence (from Richardson *et al.*, 2000).

amenity plantings, firewood, windbreaks and shade and as a fodder plant for livestock (Crompton, 1992; Le Houerou, 2000; Maslin *et al*., 2006; Maslin and McDonald., 2007; Griffin *et al*., 2011; Carruthers *et al*., 2011; Kull *et al*., 2011; Reubens *et al*., 2011). In its natural range, *A. saligna* is considered a successful farm tree for reduction of water tables and mitigation of salinity, provision of shelter and reduction in farm nutrient run-off (Bennett and George, 1993; Hobbs et al., 2009). In the semiarid Coquimbo Region, Chile, *Acacia saligna* is used particularly where **reforestation** has been promoted with the objective of recovery of degraded soils, production of fodder for livestock, fuelwood and erosion control. This alien species also has potential use as an important source of human food, because the seeds of the trees are harvested and processed for the production of breads and biscuits with nutraceutical properties (Rojas *et al*., 2016).

The primary reason for planting *A. saligna* in Libya and Ethiopia was related to the **production of fuelwood/charcoal** and as a minor uses site rehabilitation (Griffin *et al*., 2011). Over 200,000 ha of *A. saligna* have been planted in north Africa and a few thousand ha in West Asia and southeast Spain where the species is highly valued as food for sheep and goats (El-Lakany, 1987; Crompton, 1992; Le Houerou, 2002). Fuelwood may be produced at a rate of up 3.5 t dry wood 1/ha 1/year on deep sandy-loam (El-Lakany, 1987 in Midgley and Turnbull, 2003).

The phyllodes of *A. saligna* are used as a source of **fodder,** particularly for small ruminant production; the tree is often integrated into agroforestry systems in dry environments or degraded rangeland as in Kenya, Algeria (Droppelmann *et al*., 2000; Boufennara *et al*., 2013) and Chile (Meneses *et al*., 2012). However, the food intake and the digestibility of dry matter (DM), organic matter (OM) and energy contents of fresh *A. saligna* has been reported to be generally low mainly due to presence of anti-nutritional factors, such as tannins whose contents range from 47 to 55 g/kg DM. It means that the shrub could not be used as a sole dietary source for small ruminant in spite of some potential as a supplementary fodder due to its high crude protein content (Degen *et al*., 1995; Ben Salem *et al*., 1997 as reported by Tamir and Asefa, 2009).

A. saligna seeds are edible after heat treatment or cooking and can be used as a **source of human food** to combat hunger in semi-arid lands. Seeds are easily harvested and processed into flour using simple, existing local technologies; the flour can be incorporated into local dishes and in 'non-traditional' foods such as spaghetti, bread and biscuit (Rinaudo *et al*., 2002; Maslin and McDonald, 2004).

2.3 - Is the pest a vector?

YES: *Xylella fastidiosa*, a xylem-limited fastidious bacterium (EPPO A1 list, quarantine pathogen), is the recognized agent of a large number of diseases including Pierce's disease of grapevine, citrus variegated chlorosis (CVC), plum leaf scald, phony peach, pear leaf scald, alfalfa dwarf and coffee, almond, and oleander leaf scorch. Until few years ago, the presence of this bacterium was confined to the American continent, except for few sporadic reports of interception on commodities in some Asian and European countries (EFSA, 2015, 2016). As first report in the European and Mediterranean region, *X. fastidiosa* was associated to the severe olive quick decline syndrome (OQDS) in Lecce province (Apulia, southern Italy), where it is rapidly spreading (Saponari *et al*., 2013). The Apulian *X. fastidiosa* isolate was identified as a strain of the subspecies *pauca*, to which the name Codiro was assigned (Cariddi *et al*., 2014; Elbeaino *et al*., 2014)[12].

Besides olive (*Olea europaea*), *Xylella fastidiosa* subsp. *pauca* - Codiro strain can infect several other plant species, i.e., *Polygala myrtifolia*, *Westringia fruticosa*, and *Acacia saligna* (Saponari *et al*., 2013; Yaseen *et al*., 2015). Entry of the pathogen into EU territory by the movement of plants for planting is considered to be the most important pathway, since *Xylella fastidiosa* has approximately 300 reported host plant species, which include *Acacia saligna* (EFSA, 2015). Importantly, *Olea europaea* and *Acacia saligna* are very commonly closely cultivated or planted in the Mediterranean region in the European Union (e.g., Perrino and Calabrese, 2014).

[12] https://ec.europa.eu/food/sites/food/files/plant/docs/ph_biosec_legis_emergency_db-host-plants_update09.pdf

2.4 - Is a vector needed for pest entry or spread?

NO

2.5 - Regulatory status of the pest

Australia

Although this species is native only in one part of Australia, it is not declared or considered noxious by any state or territory government in Australia[13] . "*It cannot be made a proclaimed plant under the APC Act as this specifically excludes "native plants" as defined in the National Parks and Wildlife Act, 1972.*" In this latter Act the following actions are recommended: implement weed management strategies to control existing infestations and discourage the use of *A. saligna* for revegetation and landscaping (Virtue and Melland, 2003).

Europe

In **Malta**, the "Trees and Woodland Protection Regulations, 2011" (LN 200 of 2011) lists a number of species of trees deemed to cause damage to biological diversity of trees or woodlands in Malta, or to the natural environment in general. The propagation, sowing, planting, import/export, transport and selling of these 24 species (incl. *A. saligna*) are hence prohibited (MEPA 2013).

Importantly, due to the fact that besides olive (*Olea europaea*), *Xylella fastidiosa*-Codiro strain can infect *Acacia saligna* (as detailed above), there are ongoing restrictions on the movement of *A. saligna* in Europe and in the European Union. For example, in the **Republic of Montenegro**, pursuant to Article 12, paragraph 5 of the Law on Plant Health Protection ("Official Gazette of the Republic of Montenegro", number 28/06 and "Official Gazette of Montenegro", number 2 8/11 and 48/15), the Ministry of Agriculture and Rural Development passed the Order on prohibition of introduction of a list of plant (including *Acacia saligna*) for the purpose of preventing the introduction and spreading of *Xylella fastidiosa*.

In **Portugal** *Acacia saligna* is listed in the annex I of Decreto-Lei n. 565/99, of the 21st December 1999 (under the name of *Acacia cyanophylla* Lindley). This law regulates the introduction of non-native species and lists the non-native species in Portugal, indicating which are considered invasive and prohibiting the introduction of new species (with some exceptions). Furthermore, the legislation prohibits the possession, cultivation, growing and the trade of species that are considered invasive or of ecological risk.

In Cyprus, in an effort to minimise the impacts of invasive plant species on biodiversity, the Department of Forests has banned the use of known invasive species (i.e. *Acacia saligna, Ailanthus altissima, Dodonaea viscosa*) in all kinds of plantations, including those in inhabited areas and disturbed sites (Tsintides and Christou, 2011).

Israel

Acacia saligna is considered to be an invasive species in Israel and is included in a recent list of "Israel's Least Wanted Alien Ornamental Plant Species". Although this "black list" does not currently appear to have any legislative basis, it is being used by the Israel Ministry of Environmental Protection to advise planners on non-native species to avoid in planting schemes (Dufour-Dror, 2013b).

South Africa

South Africa has several regulations on invasive alien species. In particular, the art 70 of the National Environmental Management: Biodiversity Act, 2004 (Government Gazette, Republic of South Africa, Vol. 467, 7 June 2004 No. 26436) required the Minister to publish a national list of invasive species which require a range of control measures, including monitoring, removal and permits if these plants are found on private property. On the basis of the Biodiversity Act, and according to the Conservation of

[13] https://keyserver.lucidcentral.org/weeds/data/media/Html/acacia_saligna.htm

Agriculture Resources Act 1983 (Act 43 of 1983) *Acacia saligna* is listed as “CARA 2002 – Category 2 NEMBA[14] – Category 1b”[15].

[14] Invader plants may be grown under controlled conditions in permitted zones. No trade in these plants.

[15] http://www.invasives.org.za/component/k2/item/209-port-jacksons-willow-acacia-saligna - **Category 1b**: invasive species that may not be owned, imported into South Africa, grown, moved, sold, given as a gift or dumped in a waterway. Category 1b species are major invaders that may need government assistance to remove. All Category 1b species must be contained, and in many cases, they already fall under a government sponsored management programme.

2.6 - Distribution

Continent	*Distribution*	*General comments on the pest status in the different countries where it occurs according to the cited references*	*References*
Africa	Algeria	Introduced in the 1870s, widely planted/cultivated and naturalized	El Lakany (1987); Le Houerou (2000); Amrani *et al.* (2010); Boufennara *et al.* (2013); Thompson *et al.* (2015)
	Angola	Introduced, only-planted	Rejmánek *et al.* (2017)
	Botswana	Introduced, Naturalised and Invasive	Mmolotsi *et al.* (2013)
	Cape Verde	Introduced in 1988 for provenance trials	Sandys-Winsch and Harris (1992)
	Egypt	Introduced and Invasive	El Lakany (1987); El Shaer (2000); Abd El-Gawad and El-Amier (2015)
	Ethiopia	Introduced in 1870	Tamir and Asefa, (2009); Thompson *et al.* (2015)
	Kenya	Introduced around 1934, recorded still surviving in 1962 in the Nairobi Arboretum	Street (1962); Lehmann *et al.* (1999); Droppelman *et al.* (2000) as reported by Thompson *et al.* (2015)
	Libya	Introduced in 1870, widely cultivated and Naturalised, but not considered Invasive	Le Houerou (2000); Thompson *et al.* (2015)
	Morocco	Introduced, cultivated and Naturalised. By 1926 about 500,000 plants were planted to stabilise dunes near Mogador.	Jaccard (1926) as reported by Pavari and De Philippis (1941); Le Houerou (2000); Chambouleyron (2012).
	Somalia	Introduced	Bowen (1988); Thulin (1993)
	South Africa	Introduced to South Africa since 1833 and on at least five further separate occasions between 1845 and 1922, with over 200 million seeds introduced during this period. Naturalized and Invasive.	Poynton (2009) as reported by Thompson *et al.* (2011, 2015)
	Tanzania	Introduced for forest trials but not successfully established in Zanzibar with seeds from Cyprus and South Africa	Streets (1962); Kessy (1987)
	Tunisia	Introduced in the 1930s, widely cultivated and Naturalised, but not considered Invasive	Tiedeman and Johnson (1998); Le Houerou (2000); Derbel *et al.* (2009)
	Uganda	Introduced and cultivated/planned in the savannah zone and dry north-eastern lands	Dale (1953); Streets (1962)

	Zimbabwe	Introduced for reclamation of mine dumps and as ornamental	Biegel (1977); Gwaze (1987)
North America	Arizona	Introduced, only cultivated	Ebinger and Seigler (2014)
	California	Introduced and Naturalised	http://www.hear.org/pier/wra/pacific/Acacia_saligna.pdf
	Florida	Introduced, only cultivated	Atlas of Florida Plants, at: http://florida.plantatlas.usf.edu/Plant.aspx?id=4383
	Hawaii	Introduced in 1959-1960 in the Waiakea Arboretum	Richmond (1963)
Central America	Mexico	Introduced in forest trials and plantations in 1919 and in the period 1934-1940	Carabias *et al*. (2007); CONABIO (2008)
South America	Bolivia	Introduced and cultivated/planted	Killeen *et al.* (1993)
	Brazil	Introduced in 1883	Albuquerque (1889)
	Chile	Introduced in 1908, Naturalised and Invasive	Perret *et al.* (2001); Rojas *et al.* (2011); Gutierres *et al.* (2011); CABI (2017)
Asia & Middle East	Turkey	Introduced and Naturalised	Uludağ *et al.* (2017)
	Iran	Introduced and Naturalised	Irian *et al.* (2013)
	Iraq	Introduced and Invasive	Ministry of Environment, Republic of Iraq (2014)
	Israel	Introduced in 1920 and Invasive	Thompson *et al.* (2015); Cohen and Bar (2017)
	Jordan	Introduced and Invasive	Odat *et al.* (2011)
	Saudi Arabia	Introduced and Naturalised	Fadl *et al.* (2015)
Europe	Albania	Introduced and Naturalised	Rakaj *et al.* (2013)
European Union	**Croatia (EU)**	Introduced, cultivated, becoming casual	Flora Croatica Database, as reported by Giovanetti *et al.* (2014)
	Cyprus (EU)	Introduced, Naturalised and Invasive	Unwin (1926) reported by Pavari and De Philippis (1941); Streets (1962); Meikle (1977); Christodoulou (2003); Gutierres et al. (2011); Hand et al. (2011); The Administration is the civil

			government of the Sovereign Base Areas (SBBA, 2017); Pescott et al. (2018)
	France (EU) including the island of Corsica	Introduced, Naturalised and Invasive	Fried (2012); http://www.gt-ibma.eu/espece/acacia-saligna/ For Corsica: Jeanmonod (2015)
	Greece (EU) including the islands of Crete; Kithira and Rhodes	Introduced and Naturalised	Arianoutsou *et al.* (2010), *cf.* Galanos (2015) for Rhodes, for Yannitsaros (1998) for Kithira
	Italy (EU) including the islands of Sardinia & Sicily and many other small islands	Introduced since 1827 and later on widely planted for reforestation and dune stabilization (e.g. in Sardinia), Naturalised and Invasive	Maniero (2000); Celesti-Grapow *et al.* (2009, 2010); Bazan and Speciale (2002); Del Vecchio *et al.* (2013): for small Italian islands see Domina and Mazzola (2008); Celesti-Grapow *et al.* (2016)
	Malta (EU)	Introduced and Invasive	Shine *et al.* (2008)
	Portugal (EU) including Azores and Madeira	Introduced in 1869, Naturalized becoming Invasive	Gutierres *et al.* (2011); Thompson *et al.* (2015) For Madeira Menezes (1914) as reported by Da Silva Vieira (2002).
	Spain (EU) including Balearic Islands. & Canary Islands	Introduced in the XIX century, Naturalized and Invasive	San-Elorza *et al.* (2004); Gutierres *et al.* (2011); For Mallorca: http://herbarivirtual.uib.es/cas-uv/especie/4142.html For Canary Islands see, e.g., Kukel (1969); García Gallo *et al.* (2008)
Oceania	Australia (Western) Australia (New South Wales, Queensland, Tasmaania and Victoria)	Native/endemic Translocated, Naturalised and Invasive.	Maslin (1974); Virtue and Melland (2003); Maslin *et al.* (2006)
	New Zealand	Introduced and Naturalised	Heenan *et al.* (2004); Thompson *et al.* (2015); (GBIF, 2017)

2.6.1 Distribution: generalities

Acacia saligna is native (endemic) to Western Australia. It has been introduced in many other regions of the world and has naturalised mostly in Mediterranean basin, in South Africa and California (USA) (CABI, 2017). It is one of the most invasive woody species in Spain (Sanz-Elorza *et al.*, 2004), in Israel (Dufour-Dror, 2013a, b), in Cyprus and Portugal, invading sand dunes (Marchante and Marchante, 2005).

A. saligna was exported from Australia on a few occasions in the 1800s, but widespread dissemination only occurred with the formation of the Australian Tree Seed Centre in 1962 (Griffin *et al.*, 2011). The global distribution of *A. saligna* was ascertained from a wide variety of sources as reported in the table. Additional information on its distribution outside the European Union can be retrieved also from the GIASIPartnership[16] web site.

Africa

It was introduced in North Africa (e.g., in 1870 in Algeria), in other African countries and in the Middle East and largely used for stabilizing dunes, for combating desertification (Amrani *et al.*, 2010) and for agroforestry, due to its ability to thrive on sand and soils of high pH and in dry areas (Midgley and Turnbull, 2003). It is considered invasive or potentially invasive only in parts of North Africa (e.g. Algeria and Morocco) and Kenya (Thompson *et al.*, 2015). In the driest regions, such as Egypt, small plantations or trials/experimental fields are occasionally irrigated.

Acacia saligna was introduced to South Africa on at least five separate occasions between 1845 and 1922, with over 200 million seeds introduced during this period (Cronk and Fuller, 1995; Poynton, 2009; Thompson *et al.*, 2011) but it might have been introduced even earlier, around 1833, according to Cronk and Fuller (1995). It is now considered as one of the most important invasive alien plant species in the Cape Fynbos floristic region of South Africa (Thompson *et al.*, 2011, 2015).

Asia and the Middle East

Acacia saligna was introduced to many Countries both in Asia and the Middle East. The introduction of *A. saligna* from Australia into Israel was started by the British at the beginning of the twentieth century and continued by the Jewish National Fund's (JNF) forestation department for about 50 years. Due to its rapid growth rate over a broad ecological range, it was chosen for preventing soil erosion, stabilisation of mobile dunes and as a legume fodder plant in semi-arid and arid regions (Leher *et al.*, 2011). Since being planted in Israeli coastal sand dunes, *A. saligna* has spontaneously spread rapidly. This has caused significant undesired changes, from the biodiversity and conservation point of views, to the entire features of the ecosystem and to the regional biodiversity as a whole (Leher *et al.*, 2011 and reference cited therein).

Europe and the European Union

Acacia saligna was introduced in the coastal areas of several European countries (e.g., Pescotte *et al.*, 2018), mainly for sand dunes stabilisation, and for afforestation, in the Mediterranean biogeographical region. It is considered naturalised and, in many cases, also invasive, for example in sand dune habitats (e.g., Gutierres *et al.*, 2011; Arrigoni, 2010; Meloni *et al.*, 2013). The distribution for the European Union is provided in the above table (*Cf* Table 2.6). **There is available information for 8 Member States** (over 28). Importantly, the information on the presence and distribution herewith reported is in accordance with the Euro+Med PlantBase (The information resource for Euro-Mediterranean Plant Diversity)[17]. According to the available literature, **we can exclude (with low uncertainty) the presence of naturalised populations of *A. saligna* in the following 20 EU Member states**: Austria, Belgium, Bulgaria, Czech Republic, Denmark, Estonia, Finland, Germany, Hungary, Ireland, Latvia, Lithuania, Luxemburg, Netherlands, Poland, Romania, Slovakia, Slovenia, Sweden and United Kingdom. However, we cannot exclude, for these 20 countries, the presence of *A. saligna* in confined environment (Botanic Gardens, Arboreta etc.), or in forest trials or for other purposes.

In the Mediterranean region, two apparently different 'morphs' of *A. saligna* were recognized by Le Honerou (2002), i.e. an arborescent form with broad phyllodes and a form with a bushy habit and narrow phyllodes, but in the lack of further investigations these can simply be two forms of *A. saligna* subsp. *saligna*.

North, Central and South America

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[16] http://giasipartnership.myspecies.info/en

[17] http://ww2.bgbm.org/EuroPlusMed/PTaxonDetail.asp?NameId=20743&PTRefFk=8500000 [Acessed 28 October 2017].

As reported in the table, *A. saligna* has been introduced in many States in the American continent. In particular, according to Mora *et al.* (2010) the Chilean governmental agencies have projected a potential surface of more than a million hectares for plantations with this species; most of them susceptible to be covered with the Law Decree 701 for forest foster (Mora and Meneses, 2004).

Oceania

Acacia saligna is native (endemic) to Western Australia, and has been translocated to southern and eastern Australia, and is now naturalized and locally invasive from South Australia and Victoria to Queensland (Stanley and Ross, 1983).

2.7 - Habitats and where they occur in the PRA area

Habitat type (main)	EUNIS/HD habitat types	Status of habitat (e.g. threatened or protected)	Is the pest present in the habitat in the PRA area (Yes/No)	Comments (e.g. *major/minor habitats* in the PRA area)	*Reference*
Coastal habitat	B1: Coastal dunes and sandy shores (Partly threatened) Code HD 2130*: Fixed coastal dunes with herbaceous vegetation (grey dunes) Code HD 2150*: Atlantic decalcified fixed dunes (*Calluno-Ulicetea*) Code HD 2230: *Malcolmietalia* dune grasslands Code HD 2250*: Coastal dunes with *Juniperus spp* Code HD 2260: *Cisto-Lavenduletalia* dune sclerophyllous scrubs Code HD 2270*: Wooded dunes with *Pinus pinea* and/or *Pinus pinaster*	Annex I of EU Habitats Directive (92/43/EEC): 2130, 2250 and 2230. (Particularly vulnerable to disturbance and habitat modification) 2130, 2150 and 2250 are considered a priority habitat for conservation.	Yes	Common habitat type within PRA area	Gutierres *et al.* (2011); Del Vecchio *et al.* (2013); Stanisci *et al.* (2014); Farris *et al.* (2013) For Portugal: Marchante and Marchante (2005)
Heathlands **Scrub**	EUNIS F5 (Maquis, arborescent matorral and thermo-Mediterranean brushes) Code HD 5140*: *Cistus palhinhae* formations on maritime wet heaths Code HD 5220*: Arborescent matorral with *Zyziphus* Code HD 1520*: Gypsum steppes, *Gypsophiletalia* Code HD 5410; West Mediterranean clifftop phryganas (*Astragalo-Plantaginetum subulatae*)	Annex I of EU Habitats Directive (92/43/EEC): 1520, 5140, 5220 and 5410. 1520, 5140 and 5220 are considered a priority habitat for conservation.	Yes	Common habitat type in the PRA Area	Hadjikyriakou and Hadjisterkotis (2002); Fried (2010), Manolaki *et al.* (2017); For Portugal: Marchante and Marchante (2005)

Riparian wetlands and salt marshes	Code HD 1310: *Salicornia* and other annuals colonizing mud and sand Code HD 1410 Mediterranean salt meadows (*Juncetalia maritimi*) Code HD 1420: Mediterranean and thermo-Atlantic halophilous scrubs (*Sarcocornetea fruticosi*)	Annex I of EU Habitats Directive (92/43/EEC): 1310, 1410 and 1420.	Yes	Common habitat type in the PRA Area	Hadjichambis (2005); Peyton and Mountford (2015)

HD habitats (* = priority habitat): Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Codes in the table follow The Interpretation Manual of European Union Habitats - EUR 28 (April 2013)[18]. Information about the EUNIS classification can be found at: http://eunis.eea.europa.eu/about.

As summarised in the above table, a wide range of habitat types are currently invaded and threatened by *A. saligna* within the PRA area, such as coastal dunes, heatlands, scrub formations, riparian wetlands and salt marshes (see e.g Hadjikyriakou and Hadjisterkotis, 2002; Christodoulou, 2003; Gutierres *et al*., 2011; Del Veccchio *et al*., 2013; Souza-Alonso *et al*., 2017).

[18] http://ec.europa.eu/environment/nature/legislation/habitatsdirective/docs/Int_Manual_EU28.pdf

2.8 - Pathways for entry

Possible pathway	**Pathway: Plants for planting**
Short description explaining why it is considered as a pathway	*Acacia saligna* is commonly available on the market (and on-line) as seeds and live plants in pots. It is used in the PRA area as an ornamental species and for other purposes and therefore often planted also in the environment. According to the CBD terminology (UNEP/CBD/SBSTTA/18/9/Add.1, 26 June 2014) this pathway (plants for planting) can therefore be linked both to escape and release. For example (plants for planting): *http://www.murgiavivai.it/ita/piante-flora-mediterranea.asp* *http://www.jardin-du-sud.com/* *http://site.plantes-web.fr/cavatore/785/notre_histoire.htm* No documented evidence and quantitative data of recent (last 10 years) imports of *Acacia saligna* from Australia to the **European Union** was found. However, as documented by Griffin *et al.* (2011), the Australian Tree Seed Centre (ATSC) had and still has a very important role in the international dissemination of Australian acacias. The ATSC despatched samples of 322 taxa (or roughly a third of *Acacia* species native to Australia) between 1980 and 2010 to 149 countries[19]. According to Griffin *et al.* (2011), in the period **1980-2010** the ATSC despatched 29 seeds lots of *Acacia saligna* to **Europe and North America**, and 56 to the **Mediterranean region** and Middle East, thus, very likely, also to Member States of the European Union. In addition, on the web, such as in internet *fora* of garden hobbyists, in many cases, information of direct imports of seed from Australia to the European Union is found. A plethora of Australian nursery do sell on-line *Acacia saligna* seeds, for example: *https://www.nindethana.net.au/Product-Detail.aspx?p=274* *http://www.australiannativenursery.com.au/* *http://www.australianplants.com/plants.aspx?id=1501* *http://australianseed.com/shop/item/acacia-saligna* *https://www.austrahort.com.au/shop/product/233-acacia-saligna* *http://www.csiro.au/ATSCOrdering/AvailableSeedlots.aspx?SpeciesId=314*
Is the pathway prohibited in the PRA area?	In some Meber States Yes, as reported in section 2.5.
Has the pest already intercepted on the pathway?	Yes
What is the most likely stage associated with the pathway?	Seeds and plants.
What are the important factors for association with the pathway?	*Acacia saligna* is commonly available on the market (and on-line) as seeds and live plants in pots.

[19] Among those 149 countries, the following EU Member States imported *Acacia* spp. seeds: Austria, Cyprus, Belgium, Denmark, Italy, France, Germany, Hungary, Ireland, The Netherlands, Portugal, Spain, Sweden and United Kingdom.

Is the pest likely to survive transport and storage along this pathway?	Yes, seeds will easily survive transport and storage
Can the pest transfer from this pathway to a suitable habitat?	Yes. The species is often planted close to or inside natural habitats where the species can establish.
Will the volume of movement along the pathway support entry?	*Acacia saligna* is already introduced and established in significant part of the PRA area. There is only limited available information on the quantity of germplasm (mostly seeds) that is presently imported in the EU from the native range. **Importantly, very likely, and due to its old introduction, *A. saligna* is mostly propagated within the PRA area.** However, new provenances, new cultivated varieties or intra-specific hybrids might be introduced in the PRA in the near future, e.g., for bioenergy related purposes.
Will the frequency of movement along the pathway support entry?	Yes (we consider herewith "further entry" as *A. saligna* is already introduced and established in significant part of the PRA area).

Pathways for entry: Plants for planting			
Rating of the likelihood of entry for the pathway, plants or seeds for planting	**LOW**	*Moderate*	*High*
Rating of uncertainty	**LOW**	*Moderate*	*High*

2.9 - Likelihood of establishment in the natural environment in the PRA area

Acacia saligna has already established and has been described as invasive in different natural ecosystems within the Mediterranean biogeographical region of the European Union as detailed in sections 2.6-2.7, especially in **Cyprus**[20], **Italy**, **Portugal** and **Spain.** Establishment in coastal dunes, heatlands, scrub formations, riparian wetlands and salt marshes is well documented (e.g., Hadjikyriakou and Hadjisterkotis, 2002; Christodoulou, 2003; Gutierres *et al.*, 2011; Del Veccchio *et al.*, 2013; Souza-Alonso *et al.*, 2017). In addition, many LIFE projects are dedicated to *A. saligna* local eradication or control in protected areas.

Domina and Mazzola (2008) studied the ornamental flora of the islands surrounding Sicily (Italy). They reported the presence of *Acacia saligna* as cultivated species in the following islands: **Ustica**, Alicudi, **Filicudi**, Salina, Lipari, **Vulcano**, **Panarea**, **Stromboli**, Linosa, Lampedusa, **Pantelleria**, Marettimo, **Favignana** and **Levanzo**. In particular, *Acacia saligna* was recorded as naturalised over 8 of the 14 investigated islands (highlighted in bold). Similarly, Celesti-Grapow *et al*. (2016), showed that *Acacia saligna* was one of the most widespread non-native vascular plant species in a set of 37 Italians small islands, being recorded as naturalised or invasive on 16 of those islands.

The present establishment in the PRA area is due to *A. saligna* specific characteristics, such as adaptability to many environmental conditions, high seed production, large seed bank, vegetative propagation, resiliency to fires, rapid growth rates, ornamental value and many other uses that certainly promote a higher propagule pressure (Maslin and McDonald, 2004). The increase in fire frequency and intensity in the Mediterranean biogeographical region (Jolly *et al*., 2013)[21] is likely to reinforce its

[20] *Cf.* e.g., the Fourth National Report to the United Nations Convention on Biological Diversity, dated 2010, prepared by the Cyprus Department of Environment, Ministry of Agriculture, Natural Resources and Environment (https://www.cbd.int/doc/world/cy/cy-nr-04-en.pdf).

[21] According to the study of Jolly *et al.* (2013), the European Mediterranean forests are susceptible to significant changes: the inner-quartile range of fire weather season length trends indicated a lengthening of 12 to 19 days, with

populations. There is a high likelihood of further establishment in the environment in the Southern part of the European Union; it is however unlikely to establish in northern Europe because it is unlikely to grow in areas that regularly experience temperatures below freezing (Hobbs *et al.*, 2009).

Rating of the likelihood of establishment in the natural environment in the PRA area			
Rating of the likelihood of establishment in the natural environment	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	**LOW**	*Moderate*	*High*

2.10 - Likelihood of establishment in managed environment in the PRA area

Acacia saligna has also established and become invasive in managed environments within the European Union, including in tree plantations, in agricultural fields, in dunes and along road verges, where it has been planted e.g. for windbreak, soil protection and landscaping functions (Hadjikyriakou and Hadjisterkotis, 2002; Christodoulou, 2003; Guttieres *et al.*, 2011, del Vecchio *et al.*, 2013).

As for other Australian acacias, periodic soil disturbances by man from road and other infrastructure works are assisting *A. saligna*'s establishment by breaking dormancy, scaryfing the hard seed coat, providing an ideal substrate for seedling establishment and promoting re-sprouting. In managed environment, soil disturbance by man play a role similar to periodic disturbance from a natural fire regime (Spooner *et al*., 2004; Hobbs *et al*., 2009).

Rating of the likelihood of establishment in the managed environment in the PRA area			
Rating of the likelihood of establishment in the managed environment	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	**LOW**	*Moderate*	*High*

2.11 - Spread in the PRA area

2.11.1 - Natural spread

A. saligna can flower within 2-3 years and set profuse seed crops from 6 years; it is extremely fecund, with an annual seed-fall exceeding 2,000 seeds/m^2 in dense infestations (Holmes, 1990b; Virtue and Melland, 2003; McDonald *et al.*, 2007)[22]. The vast majority of seeds are rapidly shed underneath parent trees and declines rapidly when moving away from the canopy; they are adapted to dispersal by ants that carry them over a few meters and bury them in subterranean nests generating soil-stored seed banks (Milton and Hall 1981, O'Dowd and Gill, 1986; Holmes, 1990a, b; French and Major, 2001). Seeds may also be transported over longer distances by water due to buoyant pods, as highlighted by rapid invasion of riparian areas. Rodents and birds (e.g., starlings and doves) may also play some role in plant dispersal

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a maximum increase of nearly a month (29 days) from 1979 to 2013. This is consistent with a lengthening of the fire weather season in Spain during 2012 where fires burned more area than any year in the previous decade.

[22] The maximum recorded value of annual seed rain of *Acacia saligna* in the invaded range (South Africa) is 5,443 seeds/m^2 (Milton and Hall, 1981 as reported by Richardson and Kluge, 2008).

(Cronk and Fuller, 1995; Mehta, 2000; Muyt, 2001). Pods with seeds might be dispersed by wind (Danin, 2000).

A. saligna also reproduces vegetatively. Following cutting, fire and soil disturbance, it resprouts vigorously from stump and produces root suckers that could trigger the establishment of large and dense clonal stands (Virtue and Melland, 2003; Gibson *et al.*, 2011; Souza-Alonso *et al.*, 2017) [Figure 4 – Appendix 1]. However, the suckering capacity is highly dependent on subspecies. Clonal reproduction via root suckering is exhibited most strongly in *A. saligna* subsp. *stolonifera* and *A. saligna* subsp. *pruinescens*; reproduction predominantly via seed production and low propensity for root suckering are traits associated with *A. saligna* subsp. *saligna* and *A. saligna* subsp. *lindleyi* (see Table 1). As a result, there may be little evidence of clonal reproduction in some naturalised populations such as those found in the Fleurieu peninsula in South Australia originating from *A. saligna* subsp. *saligna* Eastern populations (Maslin *et al.*, 2006; McDonald *et al.*, 2007; Millar and Byrne, 2012).

2.11.2 - Human-mediated spread

The spread of *A. saligna* is strongly enhanced by both deliberate and accidental introduction by humans. Long-distance movements mostly result from intentional plantations for soil protection, amenity and the production of wood, fodder, tannin and other uses (Maslin and McDonald, 2004). Seeds and root sucker fragments are frequently transported on long distances with soil movements, wherein they can survive for long periods in a dormant stage before germinating. Human disturbance in suburban areas and along roads and railways, road works and constructions also favour species spread and local establishment (Cronk and Fuller, 1995; Muyt, 2001; Spooner *et al.*, 2004; Hobbs *et al.*, 2009; Gibson *et al.*, 2011; Wilson *et al.*, 2011; Millar and Byrne, 2012).

Importantly, as documented in the Report on the implementation of the Action Points of Recommendation No. 155 (2011) of the Standing Committee to the Bern Convention on the Illegal Killing, Trapping and Trade of Wild Birds[23], *Acacia saligna* in **Cyprus** is nowadays mainly planted by illegal bird trappers.

A. saligna is known to expand into large areas while creating homogenous landscapes (Witkowski, 1991a; Lehrer *et al.*, 2013). In Israeli coastal dunes, its cover grew by 166% over 34 years, at an annual growth rate of 2.92% which exceeds this of native vegetation; in this area, *Acacia* expansion is strongly facilitated by the exploitation of sand quarries causing topsoil movements and runoff of surface water (Bar *et al.*, 2004). In South Africa's Agulhas Plain, an active dispersion is observed from initial plantation sites to undisturbed shrublands; local regression models predicted a cover of 50% and 5% for *A. saligna*, respectively at 450 m and 5,000 m from sites of initial introduction as a result of combined effect of natural and human assisted spread (Rouget and Richardson, 2003; Yelenik *et al.*, 2004).

Where planted or established far from watercourses and in absence of human mediation, *A. saligna* seeds will not be dispersed on long distances and the plant is unlikely to spread very fast in the environement. On the contrary, a much faster spread is expected in riparian zones and because of soil movements from invaded areas. As a consequence, the overall rate of spread within the European Union is assessed as moderate.

Magnitude of spread in the PRA area			
Rating of the magnitude of spread	*Low*	**MODERATE**	*High*

[23] Council of Europe, Bern Convention, document T-PVS/Inf (2013) 13, Strasbourg, 22 July 2013, Second Conference on the Illegal killing, Trapping and Trade of Wild Birds, Tunis (31 May 2013). As reported in Scalera *et al.* (2017), *Acacia* spp. are favored by locals involved in illegal bird trapping activities (lime-sticks) due to their ability to vigorously grow and occupy an area. It is a common practice for them to plant and tend these species since they provide resting places for birds and a perfect spot for placing limesticks. Bird-trapping creates a negative image for the island abroad, with serious impact on tourism (LIFE13 NAT/CY/000176).

Rating of uncertainty	**LOW**	*Moderate*	*High*

2.12 Impact in the current area of distribution

The belief in 'miracle' plants like Australian acacias that can lift people quickly out of poverty is problematical, because such plants have the attributes of weeds - vigorous growth in degraded conditions - and often escape human control, degrading rather than improving land (Low, 2012). As described in section 2.2, Australian acacias often acquire, utilize and conserve limiting resources in invaded ecosystems better than native plants, which give them a strong competitive advantage and allows them to faster reach high size and biomass both as seedlings and as adults. Their initial high relative growth rates allow them to overtop native vegetation and outcompete natives for light that can hardly survive under its dense canopy (*A. saligna* is 123 % taller than a fynbos biome species in South Africa, *Protea repens*). Greater below-ground investment combined with mycorrhizal and N_2-fixing symbioses enables access to both water and nutrients needed to sustain growth (Witkowski, 1991b; Morris *et al.*, 2011). Another important invasive key trait of *A. saligna* is the accumulation of massive persistent seed banks in the soil that may exceed 40,000 per m^2 under tree canopy[24] and which enables it to rapidly accumulate biomass and become dominant after soil and fire disturbances promoting seed germination, thus establishing a reinforcing feedback loop that promotes its own abundance (Holmes *et al.*, 1987; Le Maitre *et al.*, 2011; Gaertner *et al.*, 2014).

A. saligna strongly impacts native biodiversity and ecosystems it invades, especially where it makes dense thickets. Negative consequences of its establishment and spread are documented from different regions in the world, mainly from South Africa where it is recognized as a major invader (Nel *et al.*, 2004), but also from Eastern Australia, Middle East and Chile (CABI, 2017).

Similarly to other Australian acacias (see Figure 1 in Appendix 3), *A. saligna* is considered as a transformer species that affects the ecosystems functions and processes as: structural and chemical soil modifications, nitrogen fixation (which provide a competitive advantage over the indigenous vegetation in the impoverished soils of the fynbos), and litter accumulation (Witkowski and Mitchell, 1987; Witkowski, 1991; Musil, 1993; Stock *et al.*, 1995; Yelenik *et al.*, 2004; Jovanovic *et al.*, 2009; Abd El Gawad and El-Amier, 2015). In general, acacias provide litter with different C-sources composition that can affect nutrient cycling and decomposition (Ens *et al.*, 2009). In particular, *A. saligna* modifies nitrogen cycling through the production of higher amounts of litter, resulting in more N being returned to the soil and an increase in the availability of inorganic nitrogen (Yelenik *et al.*, 2004).

2.12.1 - Impacts on biodiversity

The invasion of natural habitats by *A. saligna* strongly affect biodiversity. In the species-rich fynbos vegetation (shrublands) of the **Cape Floristic Region of South Africa**, tall, dense and persistent acacia stands that develop and regenerate after fire strongly reduce abundance, species richness and diversity both of the standing vegetation and the seed bank. Native species richness exhibits a marked declining trend with increasing invasion cycles; dense *A. saligna* thickets threaten endemic plant species adapted to a nutrient impoverished environment due to both shading and a strong increase of soil N, available P, pH and organic matter (Musil and Midgley, 1990; Musil, 1993; Holmes and Cowling, 1997; Richardson and van Wilgen, 2004; Yelenik *et al.*, 2004, 2007; Gaertner *et al.*, 2009; Mostert *et al.*, 2017). Areas cleared of *A. saligna* in this area hardly recover in terms of soil chemical properties and vegetation composition; the increase in soil pH and N availability favours the development of secondary invasion of weedy grasses (e.g. *Cynodon dactlylon* and *Ehrharta calycina*) and fossorial mammals after acacia stands are

[24] The maximum recorded value of seed bank of *A. saligna* in South Africa is 46,000 seeds/m2 (Holmes *et al*., 1987 as reported by Richardson and Kluge, 2008). In Cyprus, as reported in the final Report of the project LIFE12 NAT/CY/000758, several samples (25 x 25 cm) were taken from soil in three layers. The average number of seeds per square meter at the soil surface was estimated to be 1,648 seeds, at 0-10 cm depth was 2,160 seeds and at 10-20 cm was 400 seeds.

cleared for restoration purposes (Yelenik *et al.*, 2004; Holmes, 2008; Le Maitre *et al.*, 2011; Mostert *et al.*, 2017, Nsikani *et al.*, 2017). In this region, Gibson *et al.* (2012, 2013) demonstrated that prolifically flowering *A. saligna* were very attractive to honeybees and caused reduced flower visitation rate of at least one native plant species (*Roepera fulva*) with similar flowering time due to competition for pollinators whose reproductive success may be subsequently jeopardised. Its dense canopies along watercourses (35% of records in South Africa are found in riparian habitats after Morris *et al.* (2011) also shade out the habitat and threaten several species of endemic dragonflies (Samways and Taylor, 2004). Lastly, encroachment of the fynbos ecosystem by *A. saligna* affect both richness and composition of avian communities (Dures and Cumming, 2010).

Similar effects were observed in **Israeli and Egyptian coastal sand dune ecosystems** invaded by *A. saligna* spreading from nearby plantations. Invasion substantially modify the structure of vegetation cover and consequently the character of these habitats. It leads the formation of a dense cover of trees instead of an open, discountinuous, dwarf shrubs and herbaceous cover and causes a strong decrease of native plant species abundance and richness and the replacement of endemic taxa accustomed to open habitats by opportunistic species due to shading, leaf-litter accumulation, modification of soil properties and groundwater level decrease (Bar *et al.*, 2004; El-Bana, 2008; Dufour-Drop, 2012; Cohen and Bar, 2017). Invasion of coastal dunes by *A. saligna* also affects small mammal communities; the stabilization of sand dunes by the alien shrub favours human commensals such as mice and rats at the expense of the psammophile rodents (e.g. *Gerbillus pyramidum, G. andersoni allenbyi* and *Jaculus jaculus*) (Anglister *et al.*, 2005; Manor *et al.*, 2008). Similar impacts have been reported in halophytic wetlands in Cyprus (Christodoulou, 2003).

In **South Australia**, *A. saligna* is known to spread outside plantations, easily establishing amongst existing vegetation, make dense thickets, become dominant and outcompete native plants, incl. the local *Acacia pycnantha*. It is considered as an invasive weed with a very high WRA score in 4 different regions (Muyt, 2001; Melland and Virtue, 2002; Virtue and Melland, 2003).

Impact on biodiversity			
Rating of the magnitude of impact in the current area of distribution	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	**LOW**	*Moderate*	*High*

2.12.2 - Impact on ecosystem services

Acacia saligna, as other Australian acacias, is a typical example of an alien plant species that modify ecosystems and their disturbance regimes in ways that enhance their own persistence and suppress that of native species through reinforcing feedback processes (Mehta *et al.*, 2000; Gaertner *et al.*, 2014, 2017). It causes a wide range of impacts on ecosystems that increase with time and disturbance, transform habitats and originate modifications that are difficult to reverse (regime shift). It affects the delivery of ecosystem services and the benefits that society derives from them; it is known to disrupt provisioning, regulating, supporting and cultural services as demonstrated by studies performed in South African fynbos and riparian areas (e.g. Le Maitre *et al.*, 2011; Gaertner *et al.*, 2014).

In **South Africa**, several studies highlighted that economic benefits derived from the use of *A. saligna* and other Australian acacias are often exceeded by the cost of negative impacts. For example, the benefits associated to black wattle (*Acacia mearnsii*) use by commercial growers (pulp, tannin and charcoal industry) and rural users (firewood) amounted to 512 US$ million in 1998 (1 US$ = approximately 7 South African Rands) while the costs of lost streamflow (see below) are valued at 1 371 US$ million, which result in a benefit-cost ratio far below 1 (De Wit *et al.*, 2001; van Wilgen *et al.*, 2012). In comparison to *A. mearnsii*, *A. saligna* is much less planted and used by industrial growers in South Africa and in other regions of the world, the benefit-cost ratio is likely to be even lower and landowners often consider it as highly problematic. There are however two major exceptions to this general trend, where benefits typically exceed negative impacts: (i) *A. saligna* is used in its **native range** for revegetation and restoration purposes without causing substantial environemntal damage and (ii) it is also used as a multi-purpose species in arid ecosystems of **northern Africa**, where is not reported to cause adverse environmental impacts so far (Hobbs *et al*., 2009; Kull *et al*., 2011; Griffin *et al*., 2011; Wilson *et al*., 2011).

Provisioning services

The strongest documented impact of Australian acacias on ecosystem services is the reduction of both river flow (surface runoff) and groundwater recharge - termed water flows - which reduces the amount of water available for agriculture, industry and other human uses in Mediterranean areas, as well as for the flows required to sustain ecosystems downstream. Invasion in riparian habitats may even lead to complete cessation of flow during the dry season (van Wilgen *et al.*, 2008; Le Maitre *et al*., 2015; Gaertner *et al*., 2017). Due to high biomass, persistent foliage, high leaf area index and deep root system compared to native species, these invasive trees better intercept precipitation, have greater access to groundwater and have increased evapotranspiration rates which cause water flows reduction (Le Maitre *et al.*, 2000, 2011; Morris *et al.*, 2011; Catford, 2017). van Wilgen *et al.* (2008) assessed that acacias (*A. cyclops, A. longifolia, A. mearnsii, A. melanoxylon* and *A. saligna*) and other woody plants (*Eucalyptus* spp., *Hakea* spp., *Pinus pinaster* and *Prosopis glandulosa*) reduce river flow in **fynbos ecosystems** by 15% (1 064 million m^3 per year) and could potentially reduce it up to 37% (2,494 million m^3 per year) if infestation of alien plants were to reach their full potential (see graphs in Appendix 3). Similarly, alien woody plants established in riparian ecosystems in the fynbos biome cause an annual recharge reduction of groundwater aquifers of 4.4 million m^3, which can extend to 36.1 million m^3 for future levels of infestations. Depending on sources, time considered, and model used, the reduction of surface water runoff due to *Acacia saligna* alone ranges from 11.7 million m^3 to 209.9 million m^3; although being highly significant, this reduction is less than this estimated for *A. cyclops* (28.9-487.6 million m^3) and *A. mearnsii* (483.2-1077.4 million m^3), both of them covering larger areas (Le Maitre *et al.*, 2000; Le Maitre *et al.*, 2016).

Australian acacias are also known to affect other provisioning services. They have been shown to increase vegetation biomass (Milton and Siegfried, 1981; Le Maitre *et al.*, 2011), but decrease the grazing capacity of pristine vegetation in South Africa (van Wilgen *et al.*, 2008).

Regulating and supporting services

Studies in dense stands of *A. saligna* in the **South African fynbos** have documented drastic changes in litterfall dynamics and nutrient cycling leading to a strong increase in organic matter and soil and groundwater nitrogen levels (Witkowski, 1991b; Richardson and van Wilgen, 2004; Yelenik *et al.*, 2004; Jovanovic *et al.*, 2009). It has been suggested that these changes may have marked effects on fire regime and that fires will be more difficult to contain and potentially more damaging to ecosystems than fires in natural vegetation because of the strong increase of fuel loads caused by the high biomass of *A. saligna* and the relative accumulation of soil organic matter. But invasion is not likely to increase significantly fire hazard compared to native shrubland under current normal weather conditions because of lower fuel energy contents and higher moisture content of foliage; however, *A. saligna* may act to enhance fire intensity under extreme weather conditions in fynbos ecosystems, that may be favoured by climate change (i.e. air temperature > 30 °C, relative humidity < 20% and windspeed > 35 km/h) (van Wilgen and Richardson, 1985; van Wilgen and Scott 2001; Richardson and van Wilgen, 2004; Le Maitre *et al.*, 2011).

Cultural services

The presence of *A. saligna* also reduces the aesthetic and recreational quality of the fynbos due to disappearance of its beautiful ericaceous flowers which attract tourists and nature photographers (Mehta, 2000). Acacia invasion is also considered to have strongly reduced the aesthetic value of 2,000 ha of the

Nizzanim LTER nature reserve, a unique coastal dune ecosystem in Israel, and have affected tourism industry in this region (Lehrer *et al.*, 2013).

Ecosystem service (ES)	**Does the pest impact on this ES**	**Short description of impact**	**Reference**
Provisioning	Yes	Decreased diversity of fibre and food resource available, wood supply increased, water supply reduced.	Le Maitre (2000); Richardson and van Wilgen (2004); van Wilgen *et al.* (2008); Le Maitre *et al.* (2011)
Regulating and supporting	Yes	Nutrient cycling enhanced, alteration of native soil bacterial communities, microclimate altered, flood mitigation altered, habitats simplified and original ecosystem processes disrupted	Witkowski (1991b); Richardson and van Wilgen (2004); Yelenik *et al.* (2004); Jovanovic *et al.* (2009); Le Maitre *et al.* (2011); Crisóstomo *et al.* (2013)
Cultural	Yes	Recreational areas degraded and tourist experience reduced	Mehta (2000); Le Maitre *et al.* (2011); Lehrer *et al.* (2013)

Impact on ecosystem services			
Rating of the magnitude of impact in the current area of distribution	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	*Low*	**MODERATE**	*High*

2.12.3 - Socio-economic impact

The cost of invasion of **South African fynbos** shrublands by invasive woody plants is huge. It has been assessed that they have reduced the value of those ecosystems by over US$ 11.75 billion amongst which streamflow lost caused by *Acacia mearnsii* invasion amounts to US$ 1.4 billion (Higgins *et al.*, 1997; van Wilgen *et al.*, 2001). The annual loss of ecosystem services due to current level of infestation by *A. cyclops*, *A. longifolia*, *A. mearnsii* and *A. saligna* in fynbos ecosystems amounted to 210 US$ million for water provisioning, 21 US$ million for the provision of grazing for livestock and 22 US$ million for biodiversity support (data calculated from tables 3 and 4 in De Lange and van Wilgen, 2010). Unfortunately, no detailed assessment is available for the cost of *A. saligna* only regardless of the huge surfaces it covers in South Africa (i.e. 1 850 000 ha invaded in 2000, for a condensed area of 108 000 ha[25]) (Le Maitre *et al.*, 2000).

The strong hydrological impact of Australian acacias in **South Africa** (see above) led to the implementation of a highly coordinated program to control invasive alien tree called 'Working for Water'. It was initiated by the national government in 1995 to alleviate poverty (20,000 employment

[25] The condensed area is the mathematical equivalent of the total invaded area with the canopy cover adjusted to 100%.

opportunities over 15 years) and restore hydrological services by cutting down invasive woody species. Over 1.2 million hectares were cleared within the first 8 years of the program, at a yearly cost of US$ 35 million. Management costs to clear one hectare invaded by *A. saligna* including the use of fire to deplete the soil-stored seed bank are greater than the costs of 1 man-year of labour. Clearing costs of *A. saligna* in the fynbos biome incurred through the working for water program between 1995 and 2008 were valued around US$ 1 million per year (MacDonald and Wissel, 1992; van Wilgen *et al.*, 2008; van Wilgen *et al.*, 2012; Catford, 2017). The total cost of bringing invasive alien trees and shrubs under control in South Africa is estimated to be around US$ 1.2 billion, or roughly US$ 60 million per year for the estimated 20 years that it will take to deal with the problem. However, by introducing biological control as a factor, it was estimated that clearing costs over 20 years could be reduced to US$ 400 million (or US$ 20 million per year), a far more manageable target. Concerning specifically *A. saligna*, it has been assessed that the introduction of biocontrol agents since 1987 has effectively eliminated the need to proceed with expensive mechanical control programmes, yielding a return on investment of $ 800 for every $ 1 invested in the research (van Wilgen *et al.*, 2000, 2001; Impson *et al.*, 2011).

Less data concerning the socio-economic impact of *A. saligna* are available from other regions. Lehrer and Bar (2011) and Lehrer *et al.* (2013) conducted a cost-benefit analysis of the conservation management program developed to reduce the risk of *A. saligna* invasion at the Nizzanim LTER nature reserve in **Israel**. Depending on technique adopted, the total eradication treatment costs ranged from 774 to 1,590 US$ per acre; one-time cost to contain or eradicate the alien tree ranges between US$ 195,000 and US$ 400,000 which is less expensive that the annual mean willingness to pay (WTP) by visitors to protect this nature reserve.

In the **European Union** *A. saligna* is tackeld by many LIFE projects, thus a piece of information exists on control costs, e.g., LIFE08NAT/IT/000353 (€9.40 per square meter), LIFE13 NAT/IT/000433 (€17,000.00 per ha) or LIFE13 NAT/CY/000176 (€10,000.00 per ha labor cost, excluding the costs of the herbicide) (data from Scalera *et al.*, 2017), while reports from another project from Cyprus have estimated the labor cost of control at €8,630 per ha (www.care-mediflora.eu).

Among potential socio-economic impacts of *A. saligna*, it is important to take into account that this alien tree can be a host for *Xylella fastidiosa*-Codiro strain. Importantly, *Olea europaea* and *Acacia saligna* are very commonly closely cultivated or planted in the Mediterranean region in the European Union.

Finally, *A. saligna* pollen grains have shown to be allergenic in Iran, according to Irian *et al.* (2013).

Impact on socio-economics			
Rating of the magnitude of impact in the current area of distribution	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	*Low*	**MODERATE**	*High*

2.13. Potential and actual impact in the PRA area

In the European Union, *A. saligna* impacts on biodiversity mirrors the negative consequences documented in Mediterranean-type shrublands and littoral dunes of the current areas of distribution (South Africa, Middle East and Eastern Australia). Especially, sand dune ecosystems and riparian habitats are known to be invaded by large and dense thickets of the invasive shrub (i.e. the so-called ‘wattle forests’). In the **European Union** *A. saligna* is tackeld by many LIFE projects, such as LIFE13 NAT/CY/000176, LIFE13 NAT/ES/000586, LIFE08NAT/IT/000353, LIFE13 NAT/IT/000433, LIFE12 NAT/MT/000182 (data from Scalera *et al.*, 2017).

In **Cyprus**, the species has been widely planted and is currently considered amongst the most problematic invasive alien plants in the country. It creates wattle forests replacing natural vegetation and threatens several red listed plant species (e.g., *Aegilops bicornis* (Forssk.) Jaub. & Spach, *Anthemis tomentosa*, *Argyrolobium uniflorum* Jaub. & Spach, *Cladium mariscus* (L.) Pohl, *Crypsis factorovskyi* Eig, *Filago mareotica* Delile, *Isolepis cernua* (Vahl) Roem. & Schult., *Juncus maritimus* Lam., *Linum maritimum* L., *Malcolmia nana* (DC.) Boiss. var. *glabra* Meikle, *Neurada procumbens* L., *Ononis diffusa* Ten., *Tamarix hampeana* Boiss. & Heldr., Tsintides *et al*., 2007) in sand dune ecosystems but also in riparian wetlands and salt marshes on the margins of the Akrotiri and the Larnaka lakes (EC habitats 1310, 1410 and 1420) and in arborescent matorrals with *Ziziphus* (EC habitats 5220*) (Hadjikyriakou and Hadjisterkotis, 2002; Christodoulou, 2003; Hadjichambis, 2005; Delipetrou *et al.*, 2008; Peyton and Mountford, 2015; Manolaki *et al.*, 2017). Importantly, all subpopulations of the endangered plant *Aegilops bicornis* (Forssk.) Jaub. & Spach growing on sandy beaches and stabilized dunes near sea level are threatened by *A. saligna* invasion and by tourism development (Tsintides *et al*., 2007; Della *et al.*, 2007; Christou *et al.*, 2014). In addition, Lansdown et al. (2016) report the risk posed by *A. saligna* on *Callitriche pulchra* Schotsm.

In **Italy**, as a result of frequent escape from plantations established during the 1950s for reforestation/afforestation and for sand dune stabilization purposes, it forms dense monospecific stands in Italian Mediterranean dune ecosystems (especially coastal pine dune wood (EC habitat 2270*) but also Juniper dune scrublands (EC habitat 2250*) and dune sclerophyllous scrubs (EC habitat 2260*) where it favours the development of ruderal grass species at the expense of plants typical of those protected habitats (Del Vecchio *et al.*, 2013). In **Sardinia** (Italy) it outcompetes the endemic species (Endangered according to IUCN classification) *Anchusa crispa* Viv. subsp. *maritima* (Vals.) Selvi et Bigazzi (Farris *et al.*, 2013) on fixed coastal dunes with herbaceous vegetation ("grey dunes", HD 2130*). Similarly, in the island of **Sicily** (Italy), *Acacia saligna* plantations are outcompeting the endemic species *Anthyllis hermanniae* L. subsp. *brutia* Brullo et Giusso, which is Critically Endangered (according to IUCN classification, IUCN 2001, 2003, 2006) in its Sicilian type locality (*locus classicus et unicus*), as reported by Caruso (2012). A significant number of LIFE projects in Italy are locally eradicating or controlling *A. saligna* in protected areas, e.g. from the habitat 2270* (HD, Wooded dunes with *Pinus pinea* and/or *Pinus pinaster*) as in the case of the LIFE project LIFE NAT/IT/000262 “MAESTRALE”, where the presence of the non-native acacia reduces the total native diversity within the *Pinus* stands (Stanisci *et al.*, 2012), and in the Life PROVIDUNE (LIFE07NAT/IT/000519) and LIFE RES MARIS Project (LIFE13 NAT/IT/000433), both in the island of Sardinia (Italy) aiming to reduce negative impacts due to the presence of *A. saligna* in the priority habitats 2250* and 2270* (Pinna *et al.*, 2015; Acunto *et al.*, 2017). In the case of the LIFE NAT/IT/000262, the presence of *A. saligna* was shown to determine an increase of the presence of ruderal and nitrophilous species such as *Geranium purpureum* e *Oryzopsis miliacea* while reducing the presence of the species that typically characterize the dune habitats *2270 and *2250, such as *Smilax aspera* and *Pistacia lentiscus* (Calabrese *et al.*, 2017).

In **Malta**, *Tetraclinis articulata* (Regionally Endangered, IUCN) is jeopardized by habitat modification and/or destruction (including land reclamation and the clearance of the vegetation) and human-induced disturbance, including the introduction of alien species such as *Acacia saligna* and *Eucalyptus* spp. Afforestation and reforestation programmes in its distribution range with indigenous and alien trees, which do not form part of its biotope are also important threats. Competition from invasive species such as alien *Pinus* spp. and particularly the native *P. halepensis* are also seen as threats (Sánchez Gómez *et al.*, 2011).

In Sesimbra County, **Portugal**, after being introduced for afforestation purposes, *A. saligna* has proven to be very invasive in riparian habitats and sand dunes ecosystems and threatens several priority conservation habitats: fixed coastal dunes with herbaceous vegetation (EC habitat 2130*), Atlantic decalcified fixed dunes (EC habitat 2150*) and also Juniper dune scrublands (EC habitat 2250*) (Gutierres *et al.*, 2011). Crisóstomo *et al.* (2013) conducted a study to assess the diversity of symbiotic root-nodulating bacteria associated with *Acacia saligna*, in newly colonized areas in Portugal and Australia. their results supported the hypothesis that exotic *Bradyrhizobia* might have been co-introduced with *A. saligna* in Portugal. This result highlights the risks of introducing exotic inoculants that might facilitate the invasion of new areas and modify native soil bacterial communities, hindering the recovery of ecosystems.

Although no study specifically addresses the effect of *A. saligna* on ecosystem services or its socio-economic impacts within the European Union, the authors of the present PRA consider that they are similar to those documented within the current area of distribution because of similar ecological conditions and plant’s behaviour. It is also assumed that *A. saligna* has a strong effect on water provisioning services and alters water balance (i.e. soil water depletion caused by increased evapotranspiration) in coastal dune ecosystems of the Mediterranean basin, as it was shown for another invasive Australian acacia (*A. longifolia*) in the same habitat (Rascher *et al.*, 2011). Depending on invasion stage, shrub density and management objective (eradication, containment or mitigation), control costs may take very different values but is always dependent on the availability of substantial budgets (Dufour-Dror, 2013a; Reynolds, 2017).

Will impacts be largely the same as in the current area of distribution? **YES**

2.14 Identification of the endangered area

According to the climatic modelling (Appendix 4, Figure 5. a b c d) the endangered area in the European Union is composed by significant parts of the land included in the Mediterranean Biogeographical region in **Croatia**, **Cyprus**, **France**, **Greece**, **Italy**, **Malta**, **Portugal**, **Slovenia** and **Spain** and in the generality of the Mediterranean islands (with the exception of the highest mountainous regions in Sicily, Sardinia, Corsica, Crete). In addition, the endangered area includes also part of the Atlantic Region in Northern Portugal and Spain and in Western France. Part of the Continental Region in Italy is included as well. The suitability maps for the 4 *Acacia saligna* subspecies have a very similar trend and shape; however, the total size of endangered area is higher for *A. saligna* subsp. *lindleyi*, *A. saligna* subsp. *pruinescens*, *A. saligna* subsp. *stolonifera*, than in the case of *A. saligna* subsp. *saligna*. For example, the Continental region in Italy and the Atlantic region in France are very likely not at risk from the *A. saligna* subsp. *saligna* but only from the other three subspecies. The Black sea coast (**Bulgaria** and **Romania**) also appears to be marginally suitable for the establishment of the '*pruinescens*' subspecies.

The main limiting factor preventing further predicted suitability appears to be low winter temperatures. Broad habitat types at risk in the endangered area include coastland, riparian wetlands, salt marshes, heathland and scrub.

We considered in the modelling the four subspecies commonly described for *Acacia saligna*. Nevertheless, *A. saligna* subsp. *saligna* is the most important subspecies that has been commonly used as an ornamental and in re-vegetation programmes and is likely to be the subspecies most commonly utilised for agroforestry worldwide. Genetic contamination among the different genotypes are very likely to occur in the native and invasive range (Millar *et al.*, 2008a). Importantly, the genetic studies in South Africa show introduction efforts of *A. saligna* have led to an invasion that is characterized by unstructured, high genetic diversity that is divergent from that found in pure native lineages in Western Australia (Thompson *et al.*, 2012).

2.15 Climate change

Climate change is altering - and will modify also in the long run - vital aspects of the environment like temperature and precipitation, the frequency of extreme weather events, as well as atmospheric composition and land cover. The temperature, atmospheric concentration of carbon dioxide (CO_2) and available nutrients are the key factors that will drive species survival; changes in these factors will most likely stress the ecosystems and the chances of invasions (Dukes and Money, 1999; Simberloff, 2000; Dainese *et al.*, 2017). Many scientists agree that climate change will alter destination habitat and increase vulnerability to invasion because of resource scarcity and increased competition among native fauna and flora. It remains uncertain whether increasing concentrations of CO_2 in the atmosphere will generally favour non-native plant species over native plant species. Some research is suggesting that elevated CO_2 concentrations might hinder the pace of recovery of some native ecosystems after a major disturbance, like flood or fire. This could potentially lead to increased dominance of invaders in some regions (Dukes and Money, 1999).

In addition, global environmental changes could create novel environments and directly increase the availability of plant resources. Alien plants often exhibit broad environmental tolerance and high phenotypic plasticity, facilitating their successful growth in novel environments with high resource availability (Jia *et al.*, 2016 and references cited therein).

According to the **climatic projection for 2070**, the endangered area in the European Union will increase compared with the projection in the current climate (**Appendix 4, Figure 6**). The model outputs highlighted a high suitability for *Acacia saligna* s.l. in the Mediterranean Biogeographical region in Croatia, Cyprus, Italy, France, Greece, Malta, Portugal, Slovenia and Spain, and in the generality of the Mediterranean islands, as well as in the Black Sea Biogeographical region in Bulgaria and Romania. In addition, the model outputs showed a high suitability also in the Atlantic Region of Belgium, Denmark, France, Netherlands, North Germany and Southern England. Part of the Continental Region in Denmark, Poland and Boreal Region in South Sweden are included as well. The Alpine Region is unsuitable to the establishment of *A. saligna*. The suitability maps for the four *Acacia saligna* subspecies have a very similar trend and shape, however, the total size of endangered area is higher for *A. saligna* subsp. *lindleyi* and *A. saligna* subsp. *pruinescens*, than in the case of *A. saligna* subsp. *saligna* and *A. saligna* subsp. *stolonifera*. For example, for *A. saligna* subsp. *saligna* and *A. saligna* subsp. *stolonifera* in East Europe are very likely not at risk, possibly because they may be conditioned by low temperatures. On the contrary, *A. saligna* subsp. *lindleyi* and *A. saligna* subsp. *pruinescens* are likely to occupy a larger part of the Continental biogeographical region and are also predicted to be able to establish in the Pannonian biogeographical region (Hungary).

In the current climate the main limiting factor preventing further suitability appears to be low winter temperatures. Nevertheless, this factor in the future projection has been overcome, since it is shown a high suitability in colder regions. For example, *A. saligna* subsp. *lindleyi* and *A. saligna* subsp. *pruinescens*, would have in the future a high probability of establishment in Germany, Poland, Denmark and South Sweden, i.e. where the suitability was almost zero before. The 2070 model projection may underestimate the suitable range in the colder areas, since the key factor limiting spread in the EU is considered to be the severity and frequency of frosts. This may be linked to the coarse-scale modelling that does not capture local/habitat environmental conditions. Certain changes would favour *Acacia* species, however, if frosts are still likely to occur, or increase in severity and frequency, then this will more than counter any positive effects or global warming.

Important insight can be drawn for Mediterranean islands from an experiment conducted in the island of **Sardinia (Italy)** by Meloni *et al.* (2013). They showed that the optimal temperature range for germination of all populations of *A. saligna* (seeds collected in Sardinia) was 15–20 °C, but germination was also rather high at 25 °C. Increasing salt concentration influenced the germination capacity, causing a decrease in final percentages. In the presence of salt *A. saligna* germination is higher at low temperatures and it progressively decreases as the temperature increases. This is ecologically significant, in particular in coastal areas, since it indicates a need for a reduction in soil salinity for seed germination to occur, because the germination in saline environments usually occurs in spring when the temperatures are lower and soil salinity is reduced by precipitation in the late winter and spring. The investigations carried out by the Meloni *et al.* (2013) suggest, on the one hand, that the projected increase in temperatures and in summer drought length could limit the distribution of this species. On the other hand, *A. saligna* shows a tolerance to NaCl at the germination stage. *A. saligna* germination capacity is therefore one among the factors that will likely contribute, both in Sardinia and in other Mediterranean countries and territories, to an expansion of its populations in the framework of the future global change. In humid regions like Sydney, projected changes in the climate caused by atmospheric CO_2 enrichment (Clarke *et al.*, 2011) have implications for dormancy in *A. saligna* and thus its potential to develop dormant seed banks.

Finally, climate change is expected to alter the geographic distribution of wildfires, a complex abiotic process that responds to a variety of spatial and environmental gradients (Krawchuk *et al.*, 2009), a process that could promote further establishment of *Acacia saligna* close to plantations and invaded sites and may also increase species flammability and reinforce a positive feedback loop between fire disturbance and invasion (van Wilgen and Richardson, 1985; Gaertner *et al.*, 2017).

2.15.1 - Define which climate projection is being used from 2050 to 2100

Climate projection **RCP 8.5 2070**

Note: RCP[26] 8.5 is the most extreme of the RCP scenarios, and may therefore represent the worst-case scenario for reasonably anticipated climate change.

2.15.2 - Components of climate change considered most relevant for *A. saligna*

Temperature ***(YES)***	*Precipitation* ***(YES)***	*$C0_2$ levels* ***(YES)***
Sea level rise ***(NO)***	*Salinity* ***(YES)***	*Nitrogen deposition* ***(NO)***
Acidification ***(NO)***	*Land use change* ***(YES)***	

2.15.3 - Influence of projected climate change scenarios on *A. saligna*

Are the **pathways** likely to change due to climate change? **(If yes, provide a new rating for likelihood and uncertainty)**	**Reference**
The pathways are unlikely to change due to climate change	Expert opinion
Is the **likelihood of establishment** likely to change due to climate change? **(If yes, provide a new rating for likelihood and uncertainty)**	**Reference**
The likelihood of establishment is likely to increase in certain areas as a result of the increase in wildfires and winter and summer temperatures, but there is no specific evidence to support a new rating	Expert opinion; Webber *et al.* (2011); Gallardo *et al.* (2017)
Is the magnitude of **spread** likely to change due to climate change? **(If yes, provide a new rating for the magnitude of spread and uncertainty)**	**Reference**
The magnitude of spread is unlikely to change due to climate change	Expert opinion
Will **impacts** in the PRA area change due to climate change? **(If yes, provide a new rating of magnitude of impact and uncertainty for biodiversity, ecosystem services and socio-economic impacts separately)**	**Reference**
The impacts in the PRA may change due to climate change but there is no specific evidence to support a new rating	Expert opinion

[26] RCP stands for representative concentration pathways. The RCP8.5 combines assumptions about high population and relatively slow income growth with modest rates of technological change and energy intensity improvements, leading in the long term to high energy demand and GHG emissions in absence of climate change policies. Compared to the total set of Representative Concentration Pathways (RCPs), RCP8.5 thus corresponds to the pathway with the highest greenhouse gas emissions (Riahi *et al.*, 2011).

2.16 - Overall assessment of risk

Pathways for entry: Plants for planting			
Rating of the likelihood of entry for the pathway, plants or seeds for planting	**LOW**	*Moderate*	*High*
Rating of uncertainty	**LOW**	*Moderate*	*High*
Rating of the likelihood of establishment in the natural environment in the PRA area			
Rating of the likelihood of establishment in the natural environment	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	**LOW**	*Moderate*	*High*
Rating of the likelihood of establishment in the managed environment in the PRA area			
Rating of the likelihood of establishment in the managed environment	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	**LOW**	*Moderate*	*High*
Magnitude of spread in the PRA area			
Rating of the magnitude of spread	*Low*	**MODERATE**	*High*
Rating of uncertainty	**LOW**	*Moderate*	*High*
Impact on biodiversity			
Rating of the magnitude of impact in the current area of distribution	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	**LOW**	*Moderate*	*High*
Impact on ecosystem services			
Rating of the magnitude of impact in the current area of distribution	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	*Low*	**MODERATE**	*High*
Impact on socio-economics			
Rating of the magnitude of impact in the current area of distribution	*Low*	*Moderate*	**HIGH**
Rating of uncertainty	*Low*	**MODERATE**	*High*

Will impacts in the PRA area be largely the same as in the current area of distribution? **YES**

Uncertainty

Acacia saligna is a well-studied species (a large number of scientific papers are available on the Web of Science database) and has been introduced since a long time in the PRA area, where is presently described as naturalised and/or invasive in many sites, therefore the Authors would rank the uncertainty of the present PRA, in the whole document, as **LOW**.

Remarks

A significant number of other *Acacia* species (e.g., *A. dealbata* and *A. longifolia*) are present and affect biodiversity and the related ecosystem services in the European Union, therefore the Authors of the present PRA would suggest to consider them in the context of the Regulation (EU) No. 1143/2014.

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Appendix 1. Relevant illustrative pictures (for information)

Figure 1. *Acacia saligna* - inflorescences (Brundu 2017, Sardinia, IT)

Figure 2. *Acacia saligna* - glands at the base of the phyllode (Brundu 2017, Sardinia, IT)

Figure 3. *Acacia saligna* – pods and seeds (Brundu 2017, Sardinia, IT)

Figure 4. *Acacia saligna* resprouts after a wildfire (Brundu 2017, Sardinia, IT)

Figure 5. *Acacia saligna* in South Africa (Brundu 2009, South Africa)

Figure 6. *Acacia saligna* biological control in South Africa (Brundu 2009, South Africa)

Figure 7. Dense litter layer of *Acacia saligna* in Sardinia, Italy (Brundu 2017)

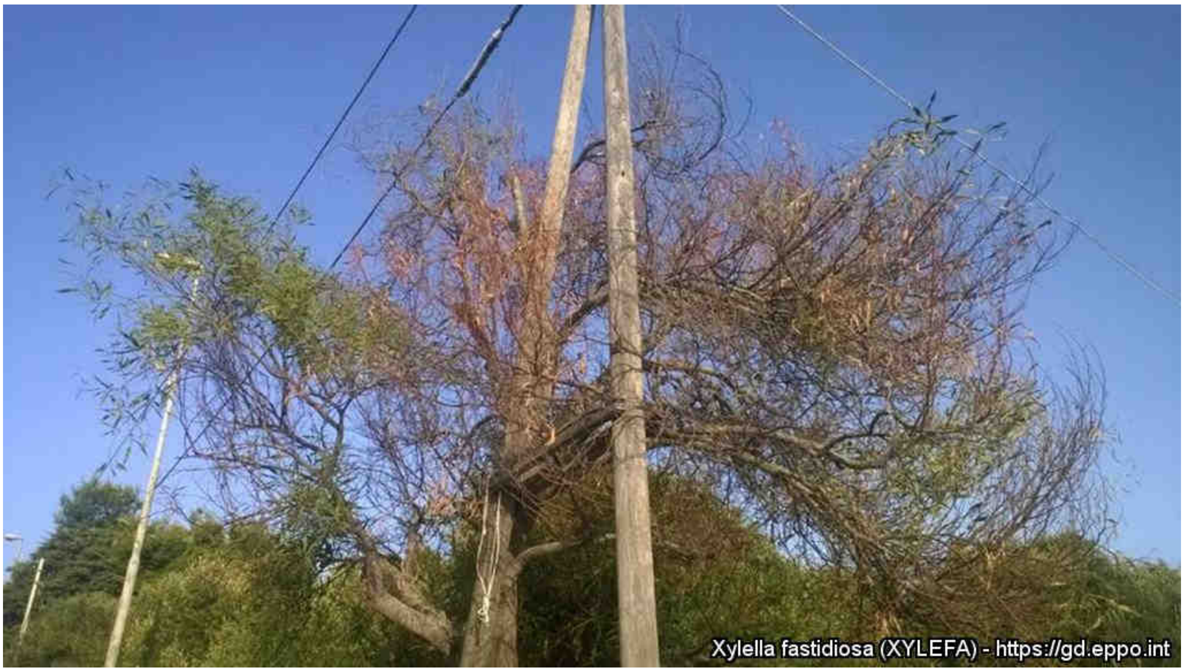

Figure 8. Courtesy of EPPO, EPPO Global database

Appendix 2. Biological traits and soil factors for *Acacia saligna* subspecies

Table 1. Biological traits and potentially undesirable attributes for the four subspecies of *Acacia saligna*, in the **native range**, as reported in the FloraBank web-site [Accessed 25 October 2017].

	Biological traits under cultivation							Potentially undesirable attributes			
***Acacia saligna* subspecies**	**Habit**	**Longevity**	**Growth rate**	**Coppicing ability**	**Root system**	**Erosion control potential**	**Carbon sequestration potential**	**Fire sensitivity**	**Foliage**	**Growth habit**	**Weediness**
A. saligna subsp. *lindleyi*	evergreen shrub < 2 m, 5 m or tree 5–10 m tall	short-lived <15 years	fast	nil or negligible	fixes nitrogen via root symbiot, forms root suckers	excellent for clayey - sandy sites	moderate- high	killed by severe fires	highly (susceptible to browsing by animals)	shallow roots may outcompete adjacent plants	declared weed or high potential
A. saligna subsp. *pruinescens*	evergreen shrub or small tree < 5 m tall	short-lived <15 years	fast	vigorous, responds to pruning	fixes nitrogen via root symbiot, forms root suckers	excellent for sandy sites	high	killed by severe fires	low - moderate (susceptibility to browsing)	shallow roots may outcompete adjacent plants	declared weed or high potential
A. saligna subsp. *saligna*	evergreen shrub or small tree < 5 m or shrub or tree 5–10 m tall	short-lived <15 years	fast	vigorous, responds to pruning	fixes nitrogen via root symbiot, forms root suckers	excellent for sandy sites	high	killed by severe fires	low - moderate (susceptibility to browsing)	shallow roots may outcompete adjacent plants	declared weed or high potential
A. saligna subsp. *stolonifera*	evergreen shrub < 2 m or shrub - small tree < 5 m tall	short-lived <15 years	fast	nil or negligible	fixes nitrogen via root symbiot, forms root suckers	excellent for sandy sites	moderate	some plants coppice back or killed by severe fires	low - moderate (susceptibility to browsing)	propensity to root sucker or shallow roots may outcompete adjacent plants	declared weed or high potential

Table 2. Soil factors and tolerances for the four subspecies of *Acacia saligna*, in the **native range**, as reported in the FloraBank web-site [Accessed 25 October 2017].

	Soil factors				**Tolerance of adverse soils**		
***Acacia saligna* subspecies**	**Texture**	**Soil pH reaction**	**Drainage**	**Salinity**	**Extremes in pH**	**Salinity (dS m-1)**	**Soil waterlogging tolerance**
A. saligna subsp. *lindleyi*	sandy, clay, loam, or sand	acidic (< 6.5) neutral (6.5–7.5)	well-drained	highly-moderately saline, or non-saline	acidity	high (9–16), moderate (–8) or slight (2–4)	nil - sensitive to waterlogged soils
A. saligna subsp. *pruinescens*	sandy, clay, loam	acidic (<6.5) neutral (6.5–7.5)	well-drained or poorly to imperfectly drained	slightly-moderately saline, or non-saline	acidity	moderate (– 8) or slight (2–4)	drainage may be sluggish at times
A. saligna subsp. *saligna*	sandy, clay, loam, or sand	neutral (6.5–7.5) or alkaline (>7.5)	well-drained	highly-moderately saline, or non-saline	alkalinity	moderate (– 8) or slight (2-4)	nil - sensitive to waterlogged soils
A. saligna subsp. *stolonifera*	sandy, clay, loam	acidic (<6.5) neutral (6.5–7.5)	well-drained	non-saline	acidity	nil - sensitive to saline soils or slight (2–4)	nil - sensitive to waterlogged soils

Appendix 3. Impact of Australian acacias on ecosystem functioning and services

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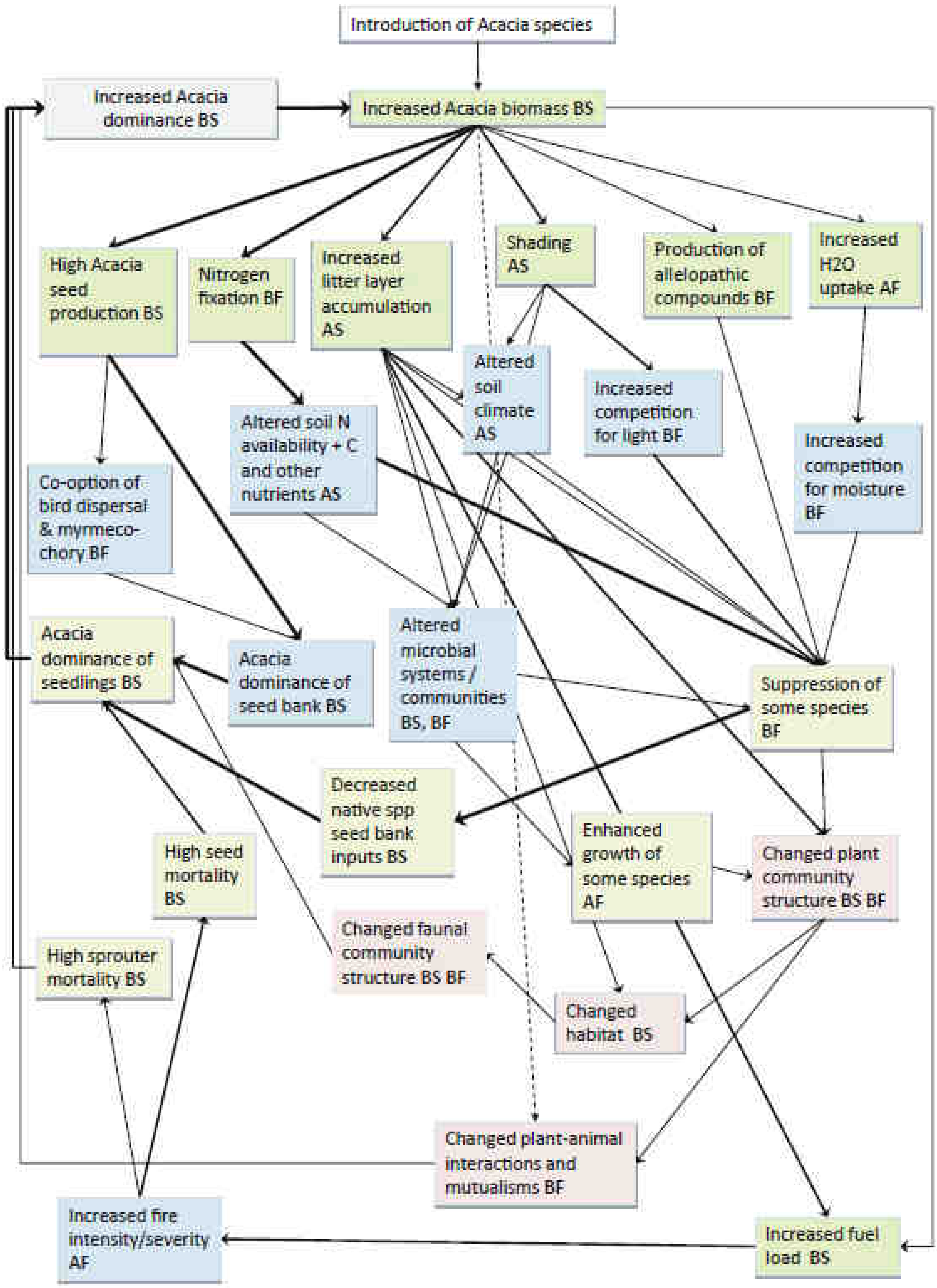

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1800 **Figure 1:** A cause-and-effect network diagram of the main impacts of Australian acacias (Le Maitre *et al.*,
1801 2011). B = biotic, A = abiotic, S = structure and F = function.

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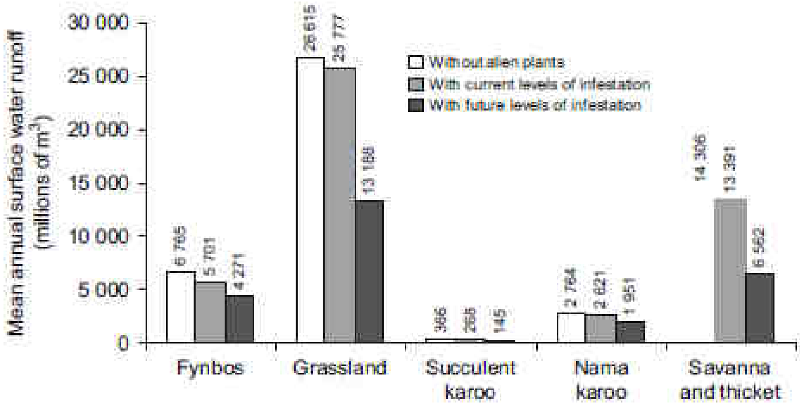


Fig. 2. Estimates of the current and potential impacts of invasive alien plants on surface water runoff in five biomes in South Africa.

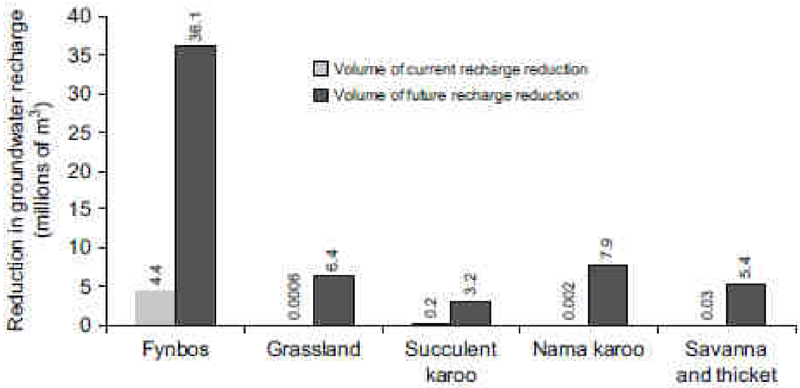


Fig. 3. Estimates of the current and potential impacts of invasive alien plants on groundwater recharge in five biomes in South Africa.

Figure 2: Effect of invasive woody species on water provisioning services in South Africa after van Wilgen *et al*. (2008).

Appendix 4. Projection of climatic suitability for *Acacia saligna* establishment

4.1 - Aim

To project the suitability for potential establishment (naturalisation) of the four subspecies of *Acacia saligna*: *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***saligna*** (autonym) '**Cyanophylla**' variant, *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***stolonifera*** M.W.McDonald & Maslin ms '**Forest**' variant, *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***pruinescens*** M.W.McDonald & Maslin ms '**Tweed River**' variant and *Acacia saligna* (Labill.) H.L.Wendl. subsp. ***lindleyi*** (Meisn.) '**Typical**' variant, in the European Union, under current and predicted future climatic conditions.

4.2 - Data for modeling

Climate data were taken from 'Bioclim' variables contained within the WorldClim database (Hijmans *et al.*, 2005) originally at 5 arcminute resolution (0.083 x 0.083 degrees of longitude/latitude) and aggregated to a 0.25 x 0.25 degree grid for use in the model. Based on the biology of the focal species, the following climate variables were used in the modelling:

- Mean minimum temperature of the coldest month (Bio6) reflecting exposure to frost. *A. saligna* subspecies exhibits frost sensitivity, and damage is likely to be severe if the temperature falls below −5 °C, suggesting this is its minimum tolerance (see climate profile in table 1).
- Mean temperature of the warmest quarter (Bio10) reflecting the growing season thermal regime. *Acacia saligna* is reported to require annual mean temperatures between 15 and 21°C under natural and cultivated conditions (see climate profile in table 2).
- Precipitation of warmest quarter (Bio18 log+1 transformed mm), also reflecting a preference for arid and semi-arid environments but not prolonged dry periods. The mean annual rainfall for the semi-arid zone is low as 300 mm (Doran and Turnbull 1997). Mean annual precipitation requirement range from 250−1200 mm, length of dry season 0-12 months (see climate profile in table 1 and 2).
- Precipitation of Coldest Quarter (Bio19 log+1 transformed mm).

The variables were also chosen based on *Acacias* modelling by Richardson *et al.* (2011) and Thompson *et al.* (2011).

To estimate the effect of climate change on the potential distribution, equivalent modelled future climate conditions for the 2070s under the Representative Concentration Pathway (RCP) 8.5 were also obtained. This assumes an increase in atmospheric CO_2 concentrations to approximately 850 ppm by the 2070s. Climate models suggest this would result in an increase in global mean temperatures of 3.7 °C by the end of the 21st century. The above variables were obtained as averages of outputs of eight Global Climate Models (BCC-CSM1-1, CCSM4, GISS-E2-R, HadGEM2-AO, IPSL-CM5A-LR, MIROC-ESM, MRI-CGCM3, NorESM1-M), downscaled and calibrated against the WorldClim baseline (see http://www.worldclim.org/cmip5_5m). RCP8.5 is the most extreme of the RCP scenarios, and may therefore represent the worst-case scenario for reasonably anticipated climate change.

In the models we also included the following variable:

- Human influence index as *A. saligna*, like many invasive species, is likely to associate with anthropogenically disturbed habitats. We used the Global Human Influence Index Dataset of the Last of the Wild Project (Wildlife Conservation Society - WCS & Center for International Earth Science Information Network - CIESIN - Columbia University, 2005), which is developed from nine global data layers covering human population pressure (population density), human land use and infrastructure (built-up areas, night-time lights, land use/land cover) and human access (coastlines, roads, railroads, navigable rivers). The index ranges between 0 and 1 and was log+1 transformed for the modelling to improve normality.

Species occurrence data were obtained from the Global Biodiversity Information Facility (GBIF), iNaturalist, USGS Biodiversity Information Serving Our Nation (BISON), Integrated Digitized Biocollections (iDigBio) and supplemented with data from the literature and from **original data collected**

by the authors of this PRA in the field in the period 2015−2017. We scrutinised occurrence records from regions where the species is not known to be well established and removed any that appeared to be dubious or where the georeferencing was too imprecise (e.g. records referenced to a country or island centroid) or outside of the coverage of the predictor layers (e.g. small island or coastal occurrences). The remaining records were gridded at a 0.25 x 0.25-degree resolution for modelling (Figure 1). Following this, there were 4490 georeferenced records and 707 grid cells with established occurrence records available for the modelling (Figure 1).

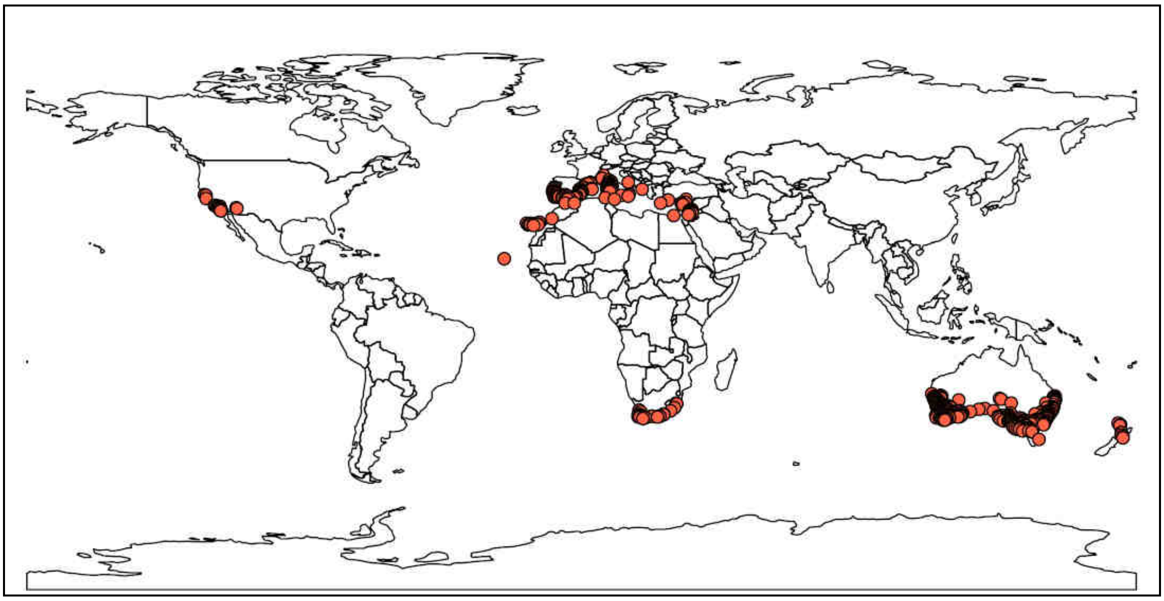


Figure 1. The selection of occurrence records of *Acacia saligna* (naturalised and casual occurrences) used in the modelling of climatic suitability in current and future climate.

Species distribution model

A presence-background (presence-only) ensemble modelling strategy was employed using the BIOMOD2 R package v3.3-7 (Thuiller *et al.*, 2009; Thuiller *et al.*, 2014). These models contrast the environment at the species' occurrence locations against a random sample of the global background environmental conditions (often termed 'pseudo-absences') in order to characterise and project suitability for occurrence. This approach has been developed for distributions that are in equilibrium with the environment. Because invasive species' distributions are not at equilibrium and subject to dispersal constraints at a global scale, we took care to minimise the inclusion of locations suitable for the species but where it has not been able to disperse to. Therefore, the background sampling region included:

- The area accessible by native *A. saligna* populations, in which the species is likely to have had sufficient time to disperse to all locations. To define the native range, we divided Australian records into native west coast populations and non-native populations on the south east. Then the accessible region was defined as a polygon bounding all native occurrences in Australia; AND
- A relatively small 25 km buffer around all non-native occurrences (including Australian ones), encompassing regions likely to have had high propagule pressure for introduction by humans and/or dispersal of the species; AND
- Regions where we have an *a priori* expectation of high unsuitability for the species (see Figure 2). Absence from these regions is considered to be irrespective of dispersal constraints. The following rules were applied to define a region expected to be highly unsuitable for *A. saligna* at the spatial scale of the model:
- Mean minimum temperature of the coldest month (Bio6). *A. saligna* is sensitive to severe frosts and the coldest occurrence has Bio6 = 0 to −5 °C suggesting this is its minimum tolerance.
- Mean temperature of the warmest quarter (Bio10). All *A. saligna* were in regions warmer than this, with the exception of a single outlying record that had Bio10 = 15 °C.

Within this sampling region there will be substantial spatial biases in recording effort, which may interfere with the characterisation of habitat suitability. Specifically, areas with a large amount of recording effort will appear more suitable than those without much recording, regardless of the underlying suitability for occurrence. Therefore, a measure of vascular plant recording effort was made by querying the Global Biodiversity Information Facility application programming interface (API) for the number of phylum Tracheophyta records in each 0.25 x 0.25-degree grid cell. The sampling of background grid cells was then weighted in proportion to the Tracheophyte recording density. Assuming Tracheophyte recording density is proportional to recording effort for the focal species, this is an appropriate null model for the species' occurrence.

To sample as much of the background environment as possible, without overloading the models with too many pseudo-absences, ten background samples of 10,000 randomly chosen grid cells were obtained (Figure 2).

Table 1. Climate profiles for the four main 'variants' described for *Acacia saligna* based on meteorological data representative of natural populations in the native range (data generated from Houlder *et al.*, 2000 and the Bureau of Meteorology website as reported by McDonald *et al.*, 2007).

Variant	Altitudinal range (m)	Mean max. hottest month (°C)	Mean min. coldest month (°C)	Lowest min. temperature recorded (°C)	Mean annual rainfall (mm)
'Typical'	100–350	28–39	5–9	– 5	250–650
'Tweed River'	150–300	30–31	4–6	– 4	700–1000
'Cyanophylla'	0–90	28–33	8–10	0	750–900
'Forest'	5–300	27–30	6–8	– 4	800–1000

Table 2. Climate profiles for the four subspecies described for *Acacia saligna* in the **native range** based on FloraBank [Accessed 25 October 2017].

	Climate parameters / tolerances				
***Acacia saligna* subspecies**	**Mean annual rainfall (mm)**	**Mean annual temperature (°C)**	**Mean max. temperature of the hottest month (°C)**	**Mean min. temperature of the coldest month (°C)**	**Frosts per year**
A. saligna subsp. *lindleyi*	250–650	15–21	28–39	5–9	up to 20
A. saligna subsp. '*pruinescens*' ms	350–1200	15–18	26–30	4–9	up to 20
A. saligna subsp. *saligna*	500–900	15–21	26–33	7–10	frost free
A. saligna subsp. '*stolonifera*' ms	800–1200	15–18	27–30	6–8	frost free

	Climate parameters / tolerances			
***Acacia saligna* subspecies**	**Frost intensity**	**Altitude (metres)**	**Drought**	**Fire**
A. saligna subsp. *lindleyi*	light–moderate (0 to – 5 °C)	100–350	moderately	killed by damaging fire
A. saligna subsp. '*pruinescens*' ms	light–moderate (0 to – 5 °C)	80–420	sensitive	killed by damaging fire
A. saligna subsp. *saligna*	light–moderate (0 to – 5 °C)	0–90	sensitive	killed by damaging fire
A. saligna subsp. '*stolonifera*' ms	light–moderate (0 to – 5 °C)	5–300	_	killed by damaging fire

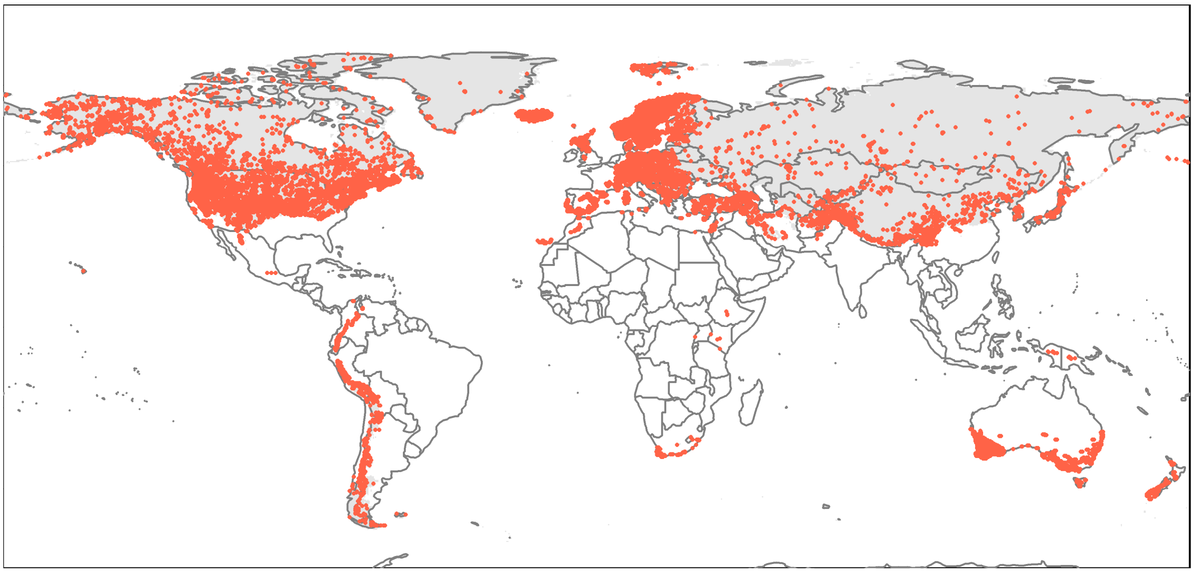


Figure 2. Randomly selected background absences in the modelling of *Acacia saligna*, mapped as red points. Points are sampled from the native range, a small buffer around non-native occurrences and from areas expected to be highly unsuitable for the species (grey background region) and weighted by a proxy for plant recording effort.

Each dataset (i.e. combination of the presences and the individual background samples) was randomly split into 80% for model training and 20% for model evaluation. With each training dataset, nine statistical algorithms were fitted with the default BIOMOD2 settings and rescaled using logistic regression, except where specified below:

- Generalised linear model (GLM)
- Generalised boosting model (GBM)
- Generalised additive model (GAM) with a maximum of four degrees of freedom per smoothing spline.
- Classification tree algorithm (CTA)
- Artificial neural network (ANN)
- Flexible discriminant analysis (FDA)
- Multivariate adaptive regression splines (MARS)
- Random forest (RF)
- MaxEnt

Since the background sample was much larger than the number of occurrences, prevalence fitting weights were applied to give equal overall importance to the occurrences and the background. Normalised variable importance was assessed and variable response functions were produced using BIOMOD2's default procedure. Model predictive performance was assessed by calculating the Area Under the Receiver-Operator Curve (AUC) for model predictions on the evaluation data, that were reserved from model fitting. AUC can be interpreted as the probability that a randomly selected presence has a higher model-predicted suitability than a randomly selected absence.

An ensemble model was created by first rejecting poorly performing algorithms with relatively extreme low AUC values and then averaging the predictions of the remaining algorithms, weighted by their AUC. To identify poorly performing algorithms, AUC values were converted into modified z-scores based on their difference to the median and the median absolute deviation across all algorithms (Iglewicz and Hoaglin, 1993). Algorithms with $z < -2$ were rejected. In this way, ensemble projections were made for each dataset and then averaged to give an overall suitability.

4.4 – Results: current climate

The ensemble model suggested that suitability for *A. saligna* was most strongly determined by the minimum temperature of the coldest month, mean temperature of the warmest quarter, and precipitation of warmest quarter (Table 1). From figure 3, the ensemble model estimated the optimum conditions for occurrence at approximately:

- Minimum temperature of the coldest month = >50% suitability for 0 - 12 °C;
- High Mean temperature of the warmest quarter;
- Low precipitation of the warmest quarter.

Precipitation of coldest quarter and Human influence index had little influence on the model predictions (Table 1, Figure 3). All these estimates are conditional on the other predictors being at their median value in the data used in model fitting.

There was substantial variation among modelling algorithms in the partial response plots (Figure 3). In part this will reflect their different treatment of interactions among variables. Since partial plots are made with other variables held at their median, there may be values of a particular variable at which this does not provide a realistic combination of variables to predict from. It also demonstrates the value of an ensemble modelling approach in averaging out the uncertainty between algorithms.

Global projection of the model in current climatic conditions indicates that the native and known invaded records generally fell within regions predicted to have high suitability (Figure 4). The model predicts potential for further expansion of the non-native range of the species into southeast Australia, south Africa, temperate and Mediterranean regions of South America, Mexico and the west coast of USA. Interestingly, several regions with unreliable records of *A. saligna* (see Figure 1) were also modelled as potentially suitable, including the east coast of USA and southeast Brazil. Elsewhere, large areas of Africa, the Middle East, India, south Asia and north Australia were projected as being potentially climatically suitable for *A. saligna* invasion (Figure 4).

The projection of suitability in Europe and the Mediterranean region suggests that *A. saligna* may be capable of establishing further populations in Portugal and southern Spain, coast of France, Italy, the Adriatic coast, Cyprus and Greece (Figure 5). There are also areas of marginal suitability predicted for coastline of North Africa (Figure 5). The main limiting factor preventing further predicted suitability appeared to be low winter temperatures.

4.5 – Results: future climate projection

According to the climatic projection in 2070, the endangered area in the European Union will increase compared with the projection in the current climate. The model includes a high suitability in the Mediterranean Biogeographical region in Croatia, Cyprus, Italy, France, Greece, Malta, Portugal, Slovenia and Spain, and in the generality of the Mediterranean islands, as well as in the Black Sea Biogeographical region in Bulgaria and Romania. The model includes a high suitability in the Atlantic Region in France, Southern England, Belgium, Netherlands and North Germany. Part of the Continental Region in Denmark is included as well. The Alpine Region is unsuitable to establishment of *A. saligna*. The suitability maps for the 4 *Acacia saligna* subspecies have a very similar trend and shape, however, the total size of endangered area is higher for *A. saligna* subsp. *lindleyi* and *A. saligna* subsp. *pruinescens*, than in the case of *A. saligna* subsp. *saligna* and *A. saligna* subsp. *stolonifera*. For example, for *A. saligna* subsp. *saligna* and *A. saligna* subsp. *stolonifera* in East Europe are very likely not at risk, possibly because they may be conditioned by low temperatures. On the contrary, *A. saligna* subsp. *lindleyi* and *A. saligna* subsp. *pruinescens* are likely to occupy a larger part of the Continental biogeographical region and are also predicted to be able to establish in the Pannonian biogeographical region (Hungary).

In the current climate the main limiting factor preventing further predicted suitability appears to be low winter temperatures. Nevertheless, this factor in the future projection has been overcome, since is shown a high suitability in colder regions. For example, for *A. saligna* subsp. *lindleyi* and *A. saligna* subsp. *pruinescens* where before the suitability was almost zero, in the future would seem an event with high probability of establishment, e.g., in **Germany**, **Poland**, **Denmark** and **South Sweden**. In this way, the 2070 model projection may underestimate the suitable range in the colder areas like mentioned before, since the key factor limiting spread in the EU is considered to be the severity and frequency of frosts. This may be linked to the coarse-scale modelling that does not capture local/habitat environmental conditions. Certain changes would favour *Acacia* species, however, if frosts are still likely to occur, or increase in severity and frequency, then this will more than counter any positive effects.

Table 3. Summary of the cross-validation predictive performance (AUC) and variable importance of the fitted model algorithms and the ensemble (AUC-weighted average of the best performing seven algorithms) for the four subspecies of *A. saligna*. Results are the average from models fitted to ten different background samples of the data.

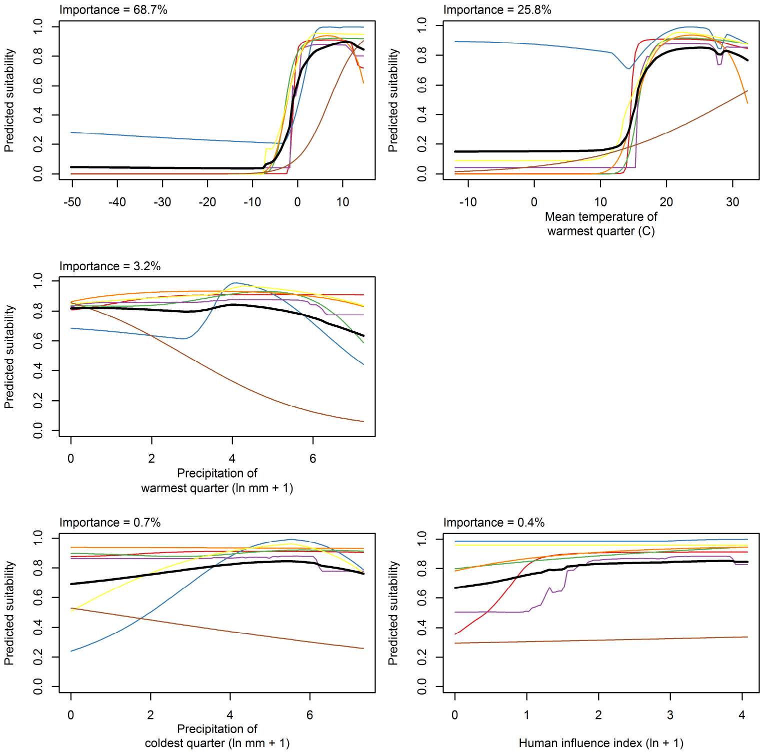
		Variable importance for *A. saligna* subsp. *lindleyi*				
Algorithm	**Predictive AUC**	**Minimum temperature of coldest month**	**Mean temperature of warmest quarter**	**Precipitation of warmest quarter**	**Precipitation of coldest quarter**	**Human Influence Index**
GLM	0.9460	66.7	33.0	0.1	0.0	0.1
GBM	0.9436	62.7	36.2	0.1	0.1	0.9
GAM	0.9502	62.9	36.8	0.2	0.0	0.1
CTA	0.9420	62.9	37.1	0.0	0.0	0.0
ANN	0.9462	62.6	32.6	1.4	0.5	1.4
FDA	0.9474	83.2	6.3	4.8	3.0	0.2
MARS	0.9470	70.9	27.9	0.4	0.5	0.0
RF	0.9072	58.6	19.4	7.9	5.1	5.1
MAXENT	0.9426	72.2	7.6	15.5	0.5	0.1
Ensemble	0.9476	68.7	25.8	3.2	0.7	0.4

		Variable importance for *A. saligna* subsp. *pruinescens*				
Algorithm	**Predictive AUC**	**Minimum temperature of coldest month**	**Mean temperature of warmest quarter**	**Precipitation of warmest quarter**	**Precipitation of coldest quarter**	**Human Influence Index**
GLM	0.9450	68.2	31.3	0.2	0.0	0.2
GBM	0.9420	63.3	35.6	0.2	0.1	0.8
GAM	0.9464	64.4	35.1	0.3	0.0	0.1
CTA	0.9396	62.9	37.1	0.0	0.0	0.0
ANN	0.9482	65.0	30.5	1.6	0.4	1.2
FDA	0.9438	84.9	5.4	4.6	2.5	0.2
MARS	0.9432	72.5	26.5	0.4	0.5	0.0
RF	0.9066	58.6	19.9	8.0	4.5	5.0
MAXENT	0.9396	73.0	7.1	15.2	0.3	0.0
Ensemble	0.9454	68.7	28.8	1.0	0.5	0.3

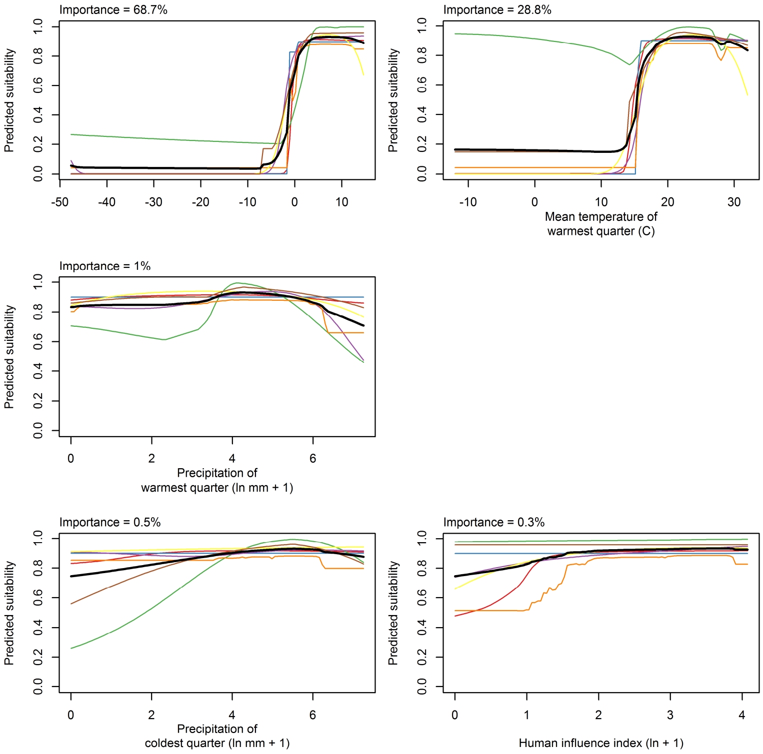
		Variable importance for *A. saligna* subsp. *saligna*				
Algorithm	**Predictive AUC**	**Minimum temperature of coldest month**	**Mean temperature of warmest quarter**	**Precipitation of warmest quarter**	**Precipitation of coldest quarter**	**Human Influence Index**
GLM	0.9504	76.2	22.6	0.7	0.0	0.0
GBM	0.9480	71.3	28.0	0.2	0.1	0.2
GAM	0.9514	74.0	25.0	0.8	0.1	0.0
CTA	0.9406	70.6	28.7	0.0	0.1	0.3
ANN	0.9506	70.5	22.6	2.8	0.7	0.6
FDA	0.9490	92.9	2.4	3.1	0.8	0.0
MARS	0.9508	79.8	19.6	0.4	0.2	0.0
RF	0.9212	66.2	14.9	7.9	3.6	3.5
MAXENT	0.9450	76.3	6.3	12.2	0.1	1.0
Ensemble	0.9500	77.3	18.1	2.9	0.3	0.3

		Variable importance for *A. saligna* subsp. *stolonifera*				
Algorithm	**Predictive AUC**	**Minimum temperature of coldest month**	**Mean temperature of warmest quarter**	**Precipitation of warmest quarter**	**Precipitation of coldest quarter**	**Human Influence Index**
GLM	0.9480	69.1	30.5	0.1	0.0	0.2
GBM	0.9448	63.9	34.7	0.1	0.1	1.0
GAM	0.9516	65.6	34.0	0.2	0.1	0.1
CTA	0.9440	63.6	36.4	0.0	0.0	0.0
ANN	0.9494	65.3	29.5	1.9	0.6	1.5
FDA	0.9484	84.8	5.6	4.5	2.5	0.2
MARS	0.9486	73.0	25.8	0.5	0.5	0.0
RF	0.9134	58.9	19.8	7.6	5.0	4.8
MAXENT	0.9444	74.0	7.4	14.2	0.6	0.0
Ensemble	0.9488	70.8	23.9	3.1	0.6	0.4

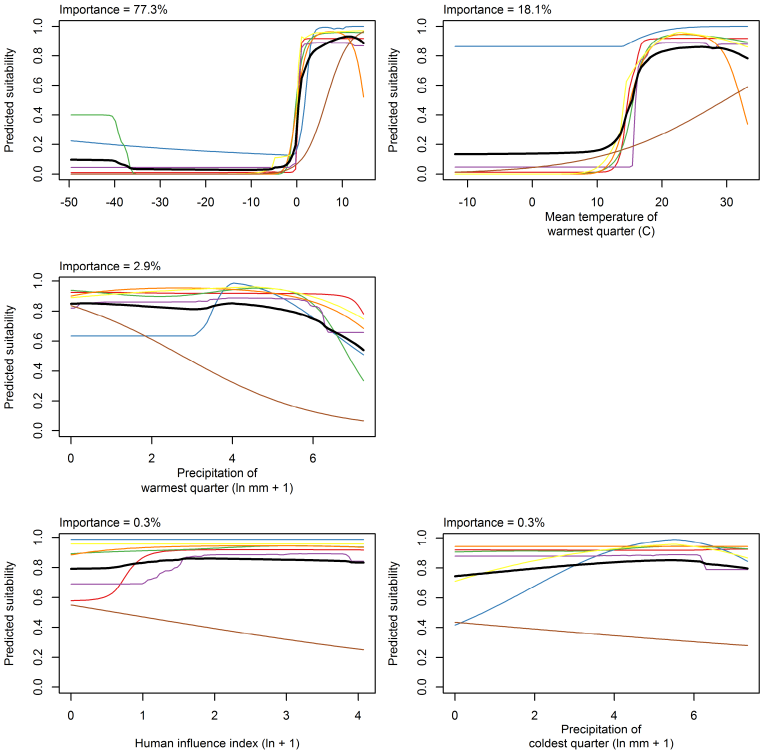
A. saligna subsp. *lindleyi*



A. saligna subsp. *pruinescens*



A. saligna subsp. *saligna* (right)



A. saligna subsp. *stolonifera*

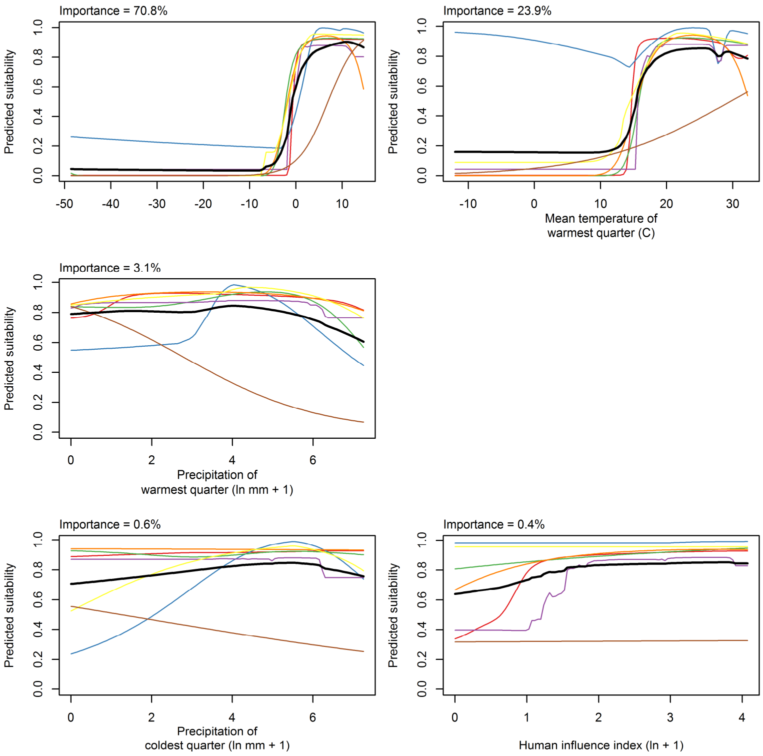
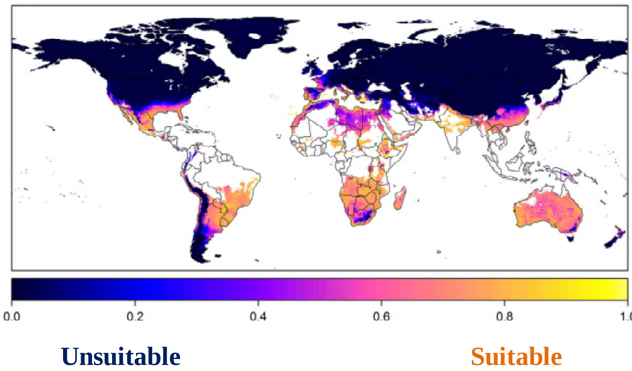
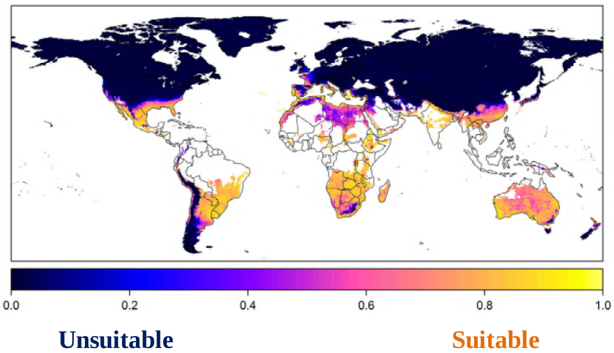


Figure 3. Partial response plots from the fitted models for the four subspecies of *A. saligna*, ordered from most to least important. Thin coloured lines show responses from the seven algorithms, while the thick black line is their ensemble. In each plot, other model variables are held at their median value in the training data. Some of the divergence among algorithms is because of their different treatment of interactions among variables.

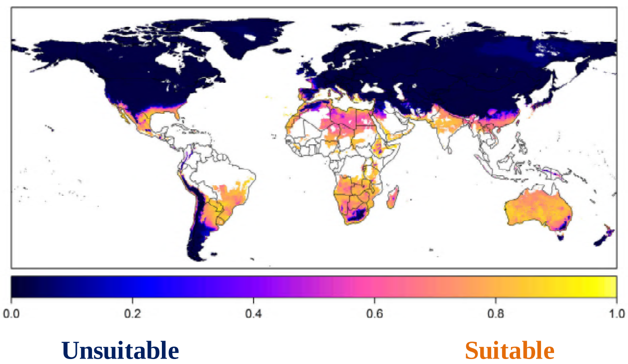
(a) *A. saligna* subsp. *lindleyi*



(b) *A. saligna* subsp. *pruinescens*



(c) *A. saligna* subsp. *saligna*



(d) *A. saligna* subsp. *stolonifera*

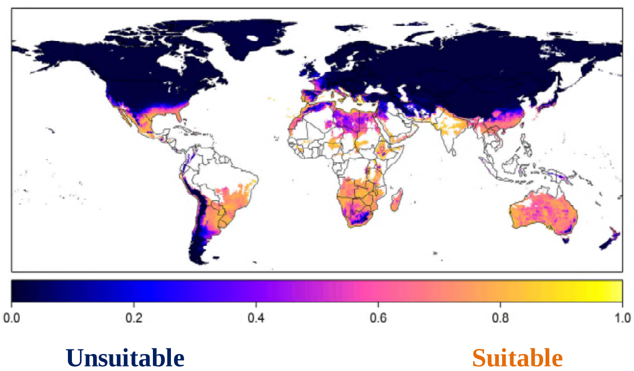


Figure 4. Projected global suitability for the four subspecies of *Acacia saligna* establishment in the current climate. For visualisation, the projection has been aggregated to a 0.5 x 0.5-degree resolution, by taking the maximum suitability of constituent higher resolution grid cells. Values > 0.5 may be suitable for the species. The white areas have climatic conditions outside the range of the training data so were excluded from the projection.

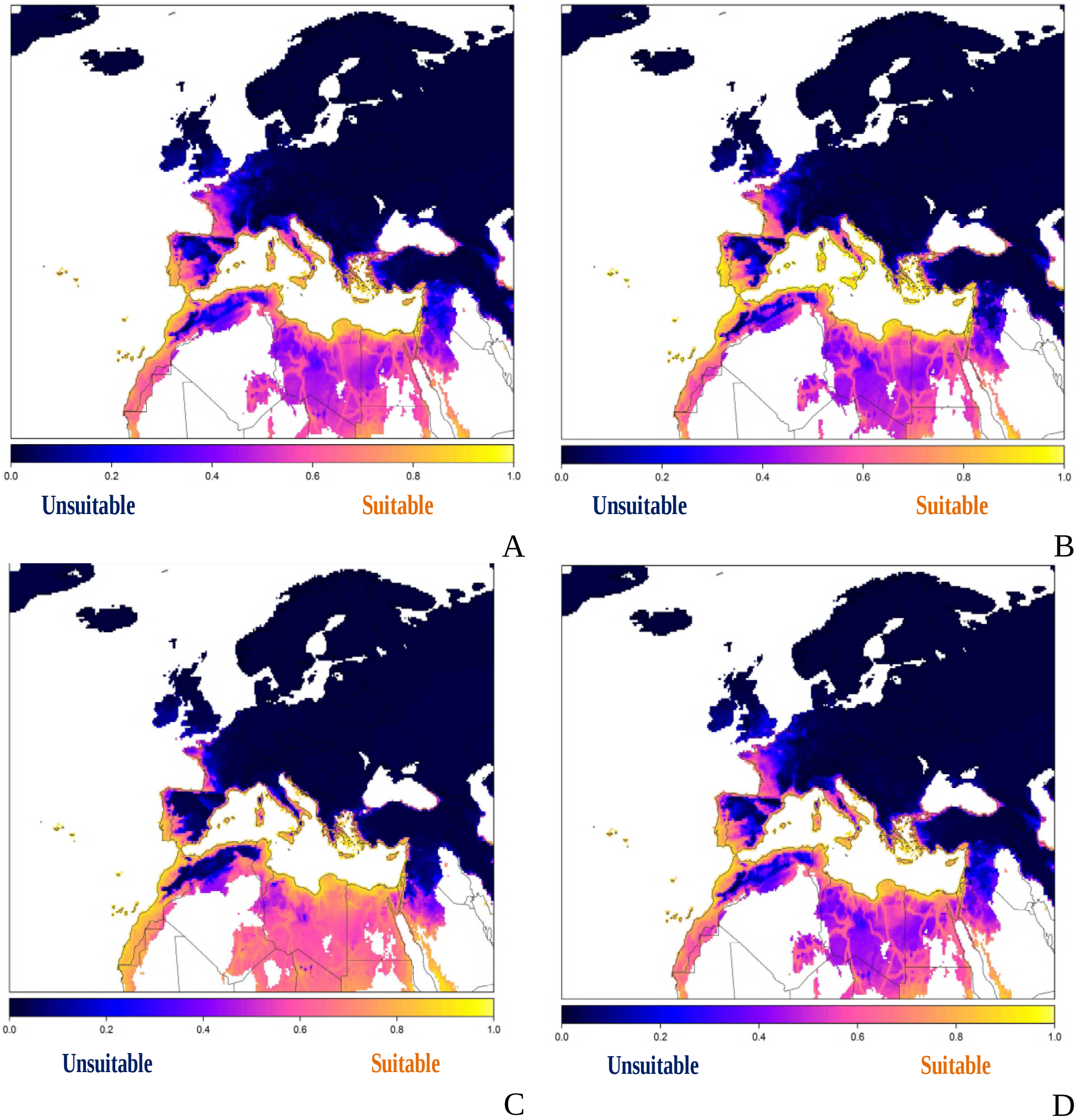


Figure 5. Projected current suitability for the four subspecies of *Acacia saligna* establishment in Europe and the Mediterranean region. The white areas have climatic conditions outside the range of the training data so were excluded from the projection. (A) *A. saligna* subsp. *lindleyi*, (B) *A. saligna* subsp. *pruinescens*, (C) *A. saligna* subsp. *saligna* and (D) *A. saligna* subsp. *stolonifera*. There are also areas of marginal suitability predicted for coastline of North Africa, as well as for the Black sea coast for the '*pruinescens*' subspecies (Bulgaria and Romania).

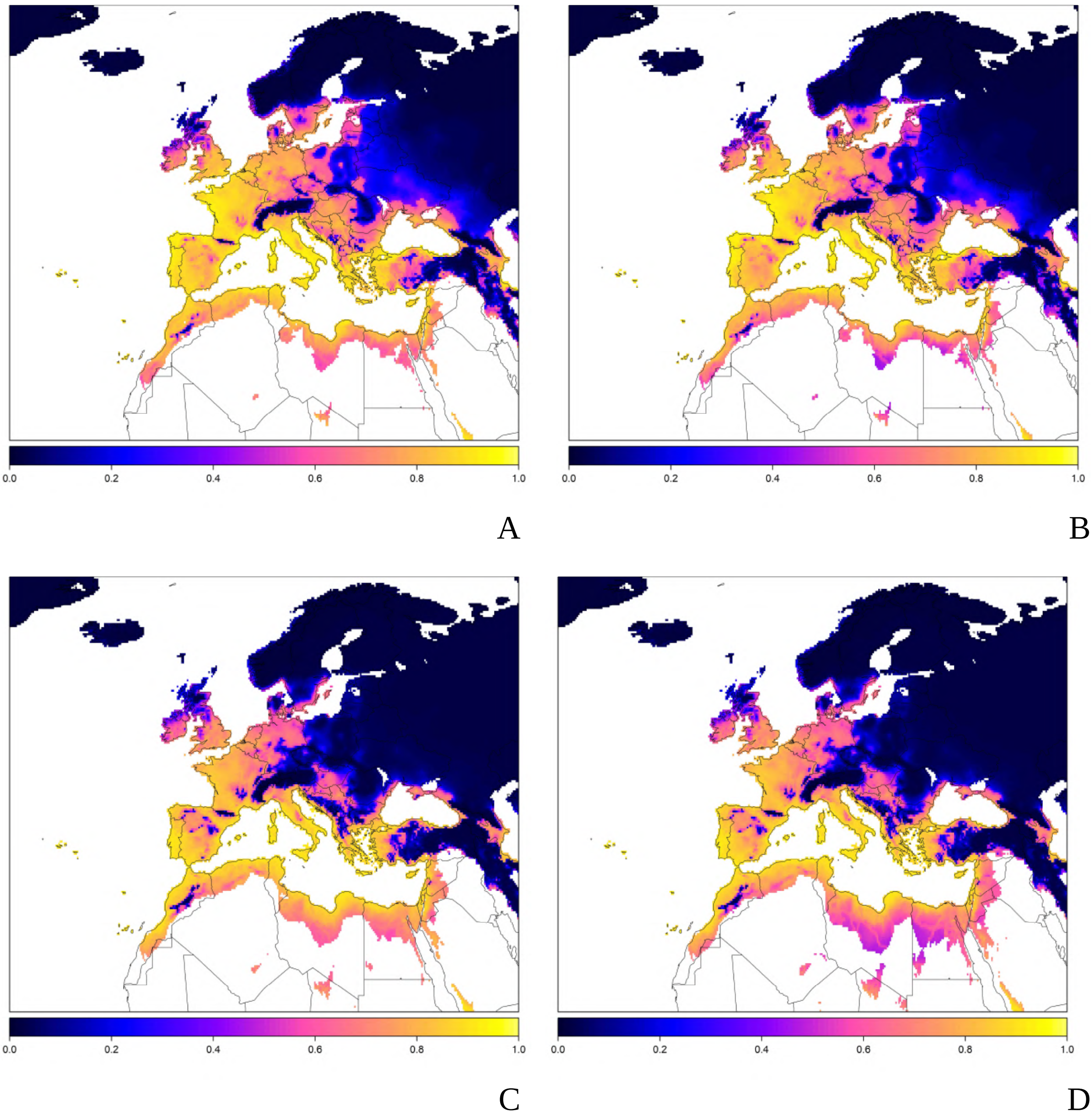


Figure 6. Projected suitability for the four subspecies of *Acacia saligna* establishment in Europe and the Mediterranean region in the 2070s under climate change scenario RCP8.5. (A) *A. saligna* subsp. *lindleyi*, (B) *A. saligna* subsp. *pruinescens*, (C) *A. saligna* subsp. *saligna* and (D) *A. saligna* subsp. *stolonifera.*

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Caveats to the modelling

There was considerable uncertainty as to the status of the *A. saligna* distribution records obtained from global databases such as GBIF. We used expert opinion to filter out records that were potentially unreliable, but it is possible that some true *A. saligna* were lost. The potential effect of this could be to underestimate the range of conditions under which the species could establish.

To remove spatial recording biases, the selection of the background sample was weighted by the density of Tracheophyte records on the Global Biodiversity Information Facility (GBIF). While this is preferable to not accounting for recording bias at all, a number of factors mean this may not be the perfect null model for species occurrence:

- The GBIF API query used to did not appear to give completely accurate results. For example, in a small number of cases, GBIF indicated no Tracheophyte records in grid cells in which it also yielded records of the focal species.
- We located additional data sources to GBIF, which may have been from regions without GBIF records.

Other variables potentially affecting the distribution of the species, such as soil nutrients or soil pH were not included in the model.

Model outputs were classified as suitable or unsuitable using a threshold of 0.5, effectively a ‘prevalence threshold’ given the prevalence weighting of model-fitting. There is disagreement about the best way to select suitability thresholds, so we evaluated the threshold selected by the commonly-used ‘minROCdist’ method. This would have selected a threshold of 0.48, slightly increasing the region predicted to be suitable.

In an expected global warming scenario with higher temperatures and CO_2 levels (IPCC 2013), with acacias growing at higher rates and producing canopies with denser foliage, reducing light availability for understory species, the invasiveness of these species could be severely increased (Souza-Alonso *et al.* 2017).