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Towards site-specific management of invasive alien trees based on the assessment of their impacts: the case of *Robinia pseudoacacia*

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Abstract

Robinia pseudoacacia L. (black locust) is a North American tree, considered controversial because of the conflict between multiple uses by humans and negative environmental impacts, which have resulted in it being listed among the most invasive species in Europe. The current management of *Robinia* stands in Central Europe varies locally according to national legislation, preferring either socio-economic benefits or biodiversity impacts.

We collected field data from our target region of Czechia, reviewed research articles including local grey literature mostly from Central and Southern Europe, unpublished results of local projects and inquired relevant specialists. Because *Robinia* grows in habitats ranging from urban to forest to natural grassland, neither unrestricted cultivation nor large-scale eradication is applicable as a universal practice. In this paper we suggest a complex management strategy for *Robinia* stands that takes into account habitat, this species' local ability to spread, as well as economic, cultural and biodiversity aspects.

We categorized *Robinia* stands growing in Europe into eight groups and proposed stratified approach to the management based on decisions that reflect local context. Depending on that, the management includes (i) establishment of new plantations, (ii) maintenance or utilization of existing stands, (iii) tolerance and (iv) conversion to original vegetation.

Our complex management strategy will provide a comprehensive guideline for the management of alien trees in Europe.

Keywords

Alien trees, *Robinia pseudoacacia*, plant invasion, nature conservation, management strategies, socio-economic benefit

Introduction

Tree species provide economic, cultural and ecological benefits to humans, often outside their native range. On the other hand, many alien trees have naturalized, subsequently become invasive and have negative environmental impacts in their introduced range. This conflict between positive and negative effects on ecosystem services poses a problem worldwide (e.g. Richardson and Rejmánek 2011, Dickie et al. 2014, Kuebbing and Simberloff 2015, Woodford et al. 2016). *Robinia pseudoacacia* is an example of such controversial tree species (Pergl et al. 2016c, Vítková et al. 2016, 2017). It is a fast growing nitrogen-fixing tree native to the south-eastern part of North America (Fowells 1965), which is planted in temperate regions worldwide (Keresztesi 1988, Li et al. 2014). Its wide utilization in native and introduced ranges started in the second half of 18th century. *Robinia* was originally planted for timber production as it is fast growing and its wood is water- and rot-resistant, and can be used as firewood or to erosion control (Vadas 1914, Göhre 1952). Large-scale afforestation campaigns were organized at the state level across Europe in the late 19th and early 20th centuries (Vítková et al. 2017). Planting and propagation of *Robinia* seemed to offer a remedy for the significant problems with deforested landscape, especially large areas of infertile pastures. Nowadays, it is the second most common broadleaved introduced tree (after *Quercus rubra*) used for forestry and wood production in Europe (MCPFE 2007). Soon after its introduction to Europe it also started to be used for amelioration, reclamation of disturbed sites, leaf forage, biomass production, honey production and shading (Papanastasis et al. 1998, Rédei et al. 2008, Yüksek 2012). Moreover, the tree is convenient for planting in urban or industrial areas, due to its tolerance of air pollution, drought, toxic, salty or nutrient-poor soils (Hillier and Lancaster 2014).

Robinia is listed among 40 most invasive woody angiosperms in the world (Richardson and Rejmánek 2011), categorized as highly invasive in several databases (EPPO, ISSG, DAISIE, CABI), ranked among the top 26 plant species in Europe with highest negative impact (Rumlerová et al. 2016) and mentioned in national Black Lists in many countries (e.g. Botta-Dukát and Balogh 2008, Celesti-Grapow et al. 2009, Vinogradova et al. 2010, Gederaas et al. 2012, Jogan et al. 2012, Seitz and Nehring 2013, Pergl et al. 2016c). The same properties that make *Robinia* attractive for cultivation are the source of problems in nature conservation and environmental management (Matus et al. 2003, Kleinbauer et al. 2010, Ivajnšič et al. 2012, Vítková et al. 2017), i.e. nitrogen fixation ability, a broad habitat tolerance, fast growth and excellent propagation ability, resulting from both prolific seed production and intensive vegetative sprouting (Batzli et al. 1992, Cierjacks et al. 2013, Vítková et al. 2015, Crosti et al. 2016).

Whereas its favourable qualities were appreciated early, the local invasions by *Robinia* started to be widely recognized only after ~1950 (Berg et al. 2016). Until then it was considered as a common naturalized tree (Hegi 1924) whose negative impacts following escape were not perceived as a problem. In traditionally deforested areas such as the Pannonian basin or Czech lowlands, *Robinia* became the main woody species planted in various habitats. It occupied the niche of local native trees, such as oaks, and replaced them in terms of importance both in the landscape and local economy. The lag between economic acceptance of *Robinia* and its rejection for impact on biodiversity took almost two centuries (Vítková et al. 2017). This period was crucial for its broad acceptance by the public. This tree became popular for its cultural value, evident from its mention in songs, poems, literature and culinary recipes (Vítková et al. 2017). Across Europe, *Robinia* is currently considered to be an integral part of the landscape and not perceived as alien by the public (Fischer et al. 2011, Lindemann-Matthies 2016). In Hungary, it is even an unofficial national tree (Keresztesi 1988). These facts demonstrate that the assessment of *Robinia* as a noxious invader needs to be balanced with its integration into landscapes and wide social acceptance.

In the last decade, the environmental and economic impacts of *Robinia* provoked stormy public debates in Europe, which involved politicians, researchers, nature conservationists, land managers, foresters, beekeepers and horticulturalists, and were recently fueled by proposal for inclusion *Robinia* on the list of invasive alien species (IAS) of Union concern (Commission Implementing Regulation 2016/1141 of 13 July 2016 pursuant to Regulation No 1143/2014 of the European Parliament and of the Council; Genovesi et al. 2015, Lehtiniemi 2016, Pergl et al. 2016a, Vítková et al. 2017), because of its impact on biodiversity, ecosystem services and human health. Unlike species with unambiguously negative environmental and/or economic impacts, *Robinia* found many defenders, who appreciated mainly its economic benefit (Tobisch and Kottek 2013). On the other hand, removing *Robinia* from the first list of perilous invaders of EU concern would compromise the ability to control this species wherever it is necessary. According to the Article 12 (the same Regulation), *Robinia pseudoacacia* may be listed in a national list of IAS of Member State concern. The control of *Robinia* invasion is even more complicated if it is not included as IAS within the legislation of the country (as in e.g. Hungary, Poland and Slovakia) and its regulation is governed by many individual enactments. Sitzia et al. (2016) highlight the potential contribution of the European forestry sector for efficient and effective implementation of EU Regulation and for controlling the spread of invasive alien species in forests. The Code of Conduct on Planted Forest and Invasive Alien Trees is voluntary and applies only to forest plantations (Brundu and Richardson 2016).

Currently, most management tools have been created for specific invaders/regions and are thus often not sufficient to address the complex range of invasion scenarios (Nielsen and Fei 2015). Our new methodological approach will provide a comprehensive guideline for the management of alien trees in Europe. We chose *Robinia pseudoacacia* as a model species because it is abundant and commonly planted, and has a great impact, both commercially and environmentally. The literature on *Robinia* is mostly one-sided, either exclusively economic or ecological. If an article deals with its utilization, it mostly lacks any consideration of the ecological problems (Rédei et al. 2008, Grünewald et al. 2009, Medinski et al. 2014), whereas if it is focused on the *Robinia* invasion, it often avoids any consideration of the economic or cultural interests (Dzwonko and Loster 1997, Kleinbauer et al. 2010, Ivajnšič et al. 2012). Here we reviewed the ecological and socio-economic impact of *Robinia* (Vítková et al. 2017) to obtain a comprehensive perspective of the invasion by this alien species in Europe. Building on the previous review (Vítková et al. 2017) we suggest a complex management strategy for *Robinia* that takes into account habitats, its ability to spread locally, as well as economic and biodiversity aspects of this invasion. Our main objectives are (i) to categorize *Robinia* populations based on their source, vegetative structure, invaded habitat, possible economic use and environmental risks, (ii) to propose site-specific management on the basis of such categorization and (iii) to compare specifics of the treatment of *Robinia* in different countries and by different stakeholders.

Material and methods

Study species

Robinia pseudoacacia L. (black locust) is a tree, but as a heliophilous and short-lived species, it is a weak competitor. This limitation is balanced by its easy and fast propagation (mainly through root suckers), tolerance of disturbance, rapid growth and tolerance of a wide range of habitats including extreme conditions. On the other hand, *Robinia* is robust and persistent, therefore it is able to persist in a site once colonized for several decades largely independent of the environment, which the tree itself modifies by changing the availability of nutrients in the soil and light conditions (Pyšek et al. 2012, Chytrý 2013, Vítková et al. 2015, Schiffleithner and Essl 2016).

Current landscape is characterized by habitat fragmentation which causes large areas of ecotones and boundary line stands, i.e. optimal conditions for *Robinia*. Serious large-scale disturbances (e.g. mining) provide a lot of open, well aerated and nutrient-rich substrata. Rotation of such disturbance events resulting in decades of successional development at abandoned sites enables *Robinia* to spread, establish and play a key role in succession. Moreover, transport of large volumes of soil containing *Robinia* propagules effectively compensates for the low ability of its large seeds to disperse over great distances.

Study area

Although most data comes from Central and Southern Europe, we considered for our assessment the whole of Europe (Table 1). Czechia (the Czech Republic) was used as the model area for the description of the management approaches as there is a lot of field data for this country (Vítková and Kolbek 2010, Vítková et al. 2015, 2016, 2017, our unpublished data) and *Robinia* is included in the Black List of IAS (Pergl et al. 2016c). We used also some data on the consequences of its planting from other parts of the world (e.g. China – Zhang 2014, Kou et al. 2016; Korea – Lee et al. 2004, Kolbek and Jarolímek 2008) to extend the applicability of suggested management strategies.

Table 1. Selected references from different European countries used for categorization and complex management strategy of *Robinia* stands. See Table 2 for description of categories indicated in the second row.

Phytosociological data	*Robinia* forests	Human-made habitats	Vulnerable habitats	Intensive short rotation plantations
	(Categories 1, 2, 3)	(Categories 4, 8)	(Categories 5, 6)	(Category 7)
Pócs (1954)	Keresztesi (1988)	Bellon et al. (1977)	Frantík (1985)	Papanastasis et al. (1998)
Jurko (1963)	Hruška (1991)	Šindelářová (1986)	Rothröckl (1986)	Platis et al. (2003)
Fekete (1965)	Benčať (1995)	Kunick (1987)	Halassy et Török (1996)	Vasilopoulos et al. (2007)
Bogojevic (1968)	LIFE99 NAT/IT/006252	Kowarik (1990, 1992, 1994, 1996)	Kelemen et Warner (1996)	Oravec (2008)
Hoff (1975)	Essl and Hauser (2003)	Swierkosz (1993)	Čechová (1998)	Rédei et al (2008)
Ščepka (1982)	Führer (2005)	Prach (1994)	Essl et Hauser (2003)	Grünewald et al. (2009)
Klauck (1988)	Novák (2005)	Sukopp and Wurzel (2003)	Matus et al. (2003)	Rédei and Veperdi (2009)
Hruška (1991)	Kalmukov (2006)	Zerbe et al. (2003)	LIFE04 NAT/CZ/000015	Kohán (2010)
Oberdorfer (1992)	LIFE08 NAT/E/000072	Kowarik and Langer (2005)	LIFE05 NAT/H/000117	Böhm et al. (2011)
Swierkosz (1993)	LIFE08 NAT/RO/000502SFC	Pietrzykowski and Krzaklewski (2006)	LIFE06 NAT/SK/000115	Rédei et al. (2011)
Kowarik and Langer (1994)	Rédei et al. (2008, 2012, 2014)	Grünewald et al. (2009)	LIFE07 NAT/B/000043	Kellezi et al. (2012)
Arrigoni (1997)	Motta et al. (2009)	LIFE11 ENV/FR/000746	LIFE07 NAT/D/000213	Stolarski et al. (2013)
Šimonovič et al. (2001)	Schneck (2010)	Yüksek (2012)	Trylč (2007)	Ciccarese et al. (2014)
Oprea (2004)	Bělař (2011)	Vlachodimos et al. (2013)	Böcker and Dirk (2007)	Medinski et al. (2014)
Benčaťová and Benčať (2005, 2008)	Essl et al. (2011)	Kanzler et al. (2015)	Bogdan (2008)	Manzone et al. (2015)
Dakskobler (2007)	Kutnar and Kobler (2013)	Wojda et al. (2015)	LIFE08 NAT/PL/000513	Wojda et al. (2015)
Willner and Grabherr (2007)	Radtke et al. (2013)	Sjöman et al. (2016)	Šefferova-Stanova et al. (2008)	Crosti et al. (2016)
Wilhalm et al. (2008)	Terwei et al. (2013)		LIFE09 NAT/IT/000118	
Campos (2010)	Ciuvăt et al. (2015)		LIFE09 NAT/CZ/000363	
Vítková and Kolbek (2010)	Malvolti et al. (2015)		Bělohlávková (2014)	
	Wojda et al. (2015)		Silva et al. (2014)	
	Akatov et al. (2016)		Schmiedel et al. (2015)	
	Budău and Timofte (2016)		Pergl et al. (2016b)	
	Sytnyk et al. (2016)		Vítková et al. (2016, 2017)	

Phytosociological data	*Robinia* forests	Human-made habitats	Vulnerable habitats	Intensive short rotation plantations
	Kadunc (2016)			
	Kou et al. (2016)			
	Schiffleithner and Essl (2016)			
	Sitzia et al. (2016)			
	Vítková et al. (2016)			

Source of data

Information for our paper, illustrating the approach for a major IAS in our study area, was obtained from (i) more than 100 research articles and local papers referring or applicable mostly to European countries (Table 1), (ii) hundreds of phytosociological relevés of *Robinia* stands growing in Europe (Table 1), (iii) inquiries addressed to European specialists (see Acknowledgments) in nature conservation, invasion ecology and management of *Robinia*, (iv) tens of results of local projects (often unpublished) testing different methods of *Robinia* removal and aftercare (e.g. Halassy and Török 1996, Novák 2005, Böcker and Dirk 2007, Trylč 2007, Bogdan 2008, Bělař 2011, Bělohlávková 2014); (v) practical experience of Czech private companies and administrations of protected areas involved in *Robinia* management, including unpublished data (e.g. Čechová 1998, Veverková 2009), and (vi) our unpublished long-term research on the ecology and impact of *Robinia* stands in various European countries. Although it might seem that there is a great body of quantitative data on, e.g. yield, growing stock, forest regeneration or eradication, in fact the available information is surprisingly poor and rather gappy. Moreover, it does not allow for comparing among individual categories of *Robinia* stands in our model area of Czechia, and even less so in other European countries. The total growing stock and yield of both planted and spontaneous *Robinia* stands could be determined only on forest land belonging to the state (not private owners) in some countries. *Robinia* stands growing on non-forest land, such as on arable land, in parks, urban and mining areas are mostly planted for other purpose than economic profit, therefore both their extent and biological parameters are not known. *Robinia* stands growing in protected areas are usually only monitored in a preparatory phase for eradication. For these reasons, it is not possible to make a rigorous statistical analysis of our general model.

Principles of the stratified approach

According to Dickie et al. (2014) we consider a dichotomy between positive and negative effects on ecosystem services resulting from planting of *Robinia* which currently causes conflicts of interest between different groups of stakeholders (e.g. nature conservation, forestry, urban landscaping, beekeepers and the public). These conflicts are often viewed only in a local context therefore we propose a complex management strategy on European level taking into account both economic benefits and environmental risks associated with *Robinia* cultivation (van Wilgen and Richardson 2014). Based on Holmes et al. (2008), Shafroth et al. (2008) and Gaertner et al. (2016), we suggest practical decision framework for sustainable *Robinia* management (Figure 1). Such framework has to be based on rigorous cost-benefit analysis (Naidoo et al. 2006, Hanley et al. 2009), leading to identification of potential conflicts. At first the potential threats associated with the presence of *Robinia* have to be identified, including threats resulting from inappropriate management of stands. If no conflict is identified, a standard management should

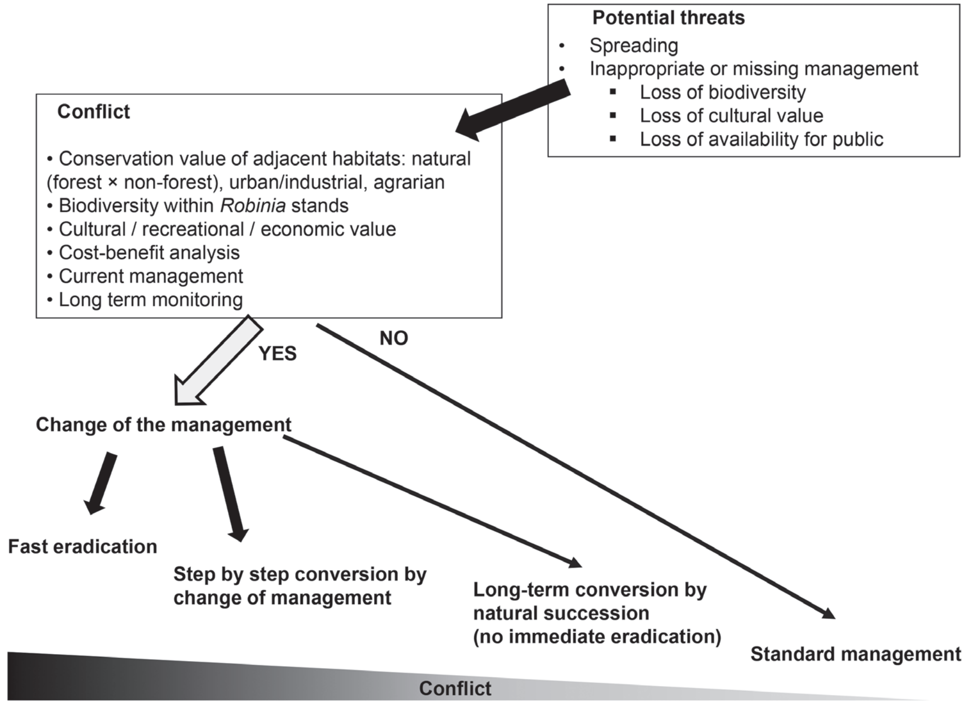

Figure 1. Decision framework for selecting suitable *Robinia* management. Width of arrows indicate importance of the management. Shading indicates the number of potential sites covered (white – relatively few occurrences, black – most of the sites). Data come from the reviewed literature and project reports.

continue (management of plantations, ornamental trees). In presence of any conflict the recommended management depends on the intensity of the threat ranging from slow conversion by succession to fast eradication. In addition, the decision scheme needs to be accompanied by categorization of stands with *Robinia* into eight groups (Table 2) reflecting the variation in habitat conditions and character of stands, in order to make context-dependent decisions relevant to local conditions. For each group, the distribution and source of *Robinia*, its history, ecological characteristics (habitat, structure, plant composition) and currently used management are summarized.

Results and discussion

Categorization of *Robinia* stands according to their management and impact

Based on links between ecological traits such as habitat, vegetation structure, origin, utilization, benefits and environmental risks we distinguish eight types of *Robinia* stands (Table 2). Each type includes four management practices, which are effective in various combinations depending on local conditions: (i) establishment of stands,

Table 2. Main features used in categorization of types of *Robinia* stands, their description and management.

<table>
<tr><th>Robinia type</th><th>Physiognomy</th><th>Distribution and habitats</th><th>Source of occurence</th><th>Vegetation structure and dynamics</th><th>Status</th><th>Management</th></tr>
<tr><td rowspan="3">1. Regularly managed Robinia forests</td><td rowspan="8">Closed forests in natural habitats</td><td>Common, temperate and warm areas across Europe</td><td rowspan="2">Cultural (open habitats on initially infertile soils threatened by soil erosion, mainly sands and rocky pastures)</td><td>Monospecific tree layer</td><td rowspan="3">Sustainable - profitable - risky</td><td>Forestry maintenance</td></tr>
<tr><td rowspan="2">Wide range of habitats (from wind-blown nutrient poor sands to the most fertile soils)</td><td>Dense and species-rich undergrowth dominated by nitrophilous herbs or grasses</td><td rowspan="2">Conversion is troublesome and risky</td></tr>
<tr><td>Spontaneous (open habitats in the vicinity of plantations, e.g. abandoned vineyards, orchards and fields)</td><td>Regular regeneration (forestry)</td></tr>
<tr><td rowspan="3">2. Regularly managed mixed Robinia forests</td><td>The most common forest type with Robinia</td><td>Cultural (open woodlands, gappy forests, clearings, deforested sites)</td><td>Mixed - Robinia combines with alien and native trees</td><td rowspan="3">Sustainable - profitable - low risk</td><td>Forestry maintenance</td></tr>
<tr><td rowspan="2">Wide range of habitats across Europe</td><td rowspan="2">Spontaneous (drier parts of floodplain forests, forest margins, disturbed sites)</td><td>Biodiversity of undergrowth depends on the share of Robinia and other aliens</td><td rowspan="2">Conversion is troublesome and risky, succession to natural forests is easy</td></tr>
<tr><td>Regular regeneration (forestry)</td></tr>
<tr><td rowspan="2">3. Unmanaged old Robinia forests</td><td>Czechia, Switzerland</td><td>Abandoned old cultures (over 50 years)</td><td>Mixed - Robinia gradually replaced by competitive trees (Fraxinus excelsior, Acer platanoides)</td><td rowspan="2">Instable - not profitable - risky</td><td rowspan="2">Conversion is troublesome and risky, succession to natural forests is easy</td></tr>
<tr><td>Less accessible sites, e.g. steep slopes</td><td>Spontaneous old and never managed stands, e.g. in rocky ravines</td><td>Spontaneous succession without management for several decades</td></tr>
</table>

Robinia type	Physiognomy	Distribution and habitats	Source of occurence	Vegetation structure and dynamics	Status	Management
4. Stands in human-made habitats	Small-scale or semi-open non-forest stands	Common, widespread across Europe	Cultural (ornamental purposes, apiculture or biological recultivation)	Monospecific or mixed stands with native pioneers, nitrophilous trees and aliens	Sustainable - context dependent risk and profit	Context dependent planting or conversion
		Urban, agrarian and industrial areas	Spontaneous (escape and succession in wasteland, public greenery, post-mining landscape and landfills)	High share of aliens in canopy or understory; many ornamental woody species		
		Mining areas		Ruderal undergrowth; diverse dynamic		
5. Dwarf Robinia stands growing in natural grasslands		Pannonian lowland, South and South-East Europe	Unsuccesfull cultivation combined with spontaneous spread	Low, twisted trees (ca 5-10m) or shrubs with native xerophilous shrubs	Sustainable - not profitable - low risk	Conversion / removal is troublesome, risky and mostly not necessary
		Dry habitats (mostly mosaic of grassland, shrubs and open woodland)		Many species of sunny open habitats; nitrophytes are rare due to drought		
				Stable stands; survival of rare species preserves local biodiversity in agricultural land		
6. Young *Robinia* stands spreading into vulnerable habitats		Dry to mesic grasslands across Europe	Spontaneous spread by root suckers from adjacent stands	Young shoots with increasing cover	Instable - not profitable - risky	Conversion / removal is troublesome and risky but necessary
		Rare vulnerable native habitats		Valuable herbaceous vegetation is replaced by *Robinia*		
				Instable stands		

Robinia type	Physiognomy	Distribution and habitats	Source of occurence	Vegetation structure and dynamics	Status	Management
7. Intensive short rotation plantations	Biomass cultures	Across South and Central Europe – e.g. Albania, Austria, Italy, Germany, Greece, Hungary, Poland, Slovakia and Spain	Cultural	Monospecific, low height (average 5-6m), rapid regeneration	Instable - profitable - risky	Intensive cultivation
		Both in forests and arable land		Low biodiversity value, weeds or nitrophytes prevail		Conversion of abandoned plantations to forest
8. Cultivated single trees and avenues	Separate tree individuals	Common across Europe	Cultural	Horticultural treatment, protection of old monument trees	Sustainable - context dependent risk and profit	Context dependent planting or conversion

(ii) maintenance of the existing state or utilization, (iii) tolerance of natural succession without major human interventions, and (iv) conversion, i.e. management or measures targeted at changing a stand into another unit or type of vegetation. The advantages and risks of particular management practices are discussed in the context of different initial conditions. Relations among the types of *Robinia* stands distinguished, successional trends and suitable management practices are shown in Figure 2.

1. Regularly managed *Robinia* forests (Table 2, Figures 2, 3A)

Deep, well-aerated, nutrient-rich mesic soils in warm areas are optimal for the growth of *Robinia* since trees reach up to 35 m, form straight trunks and provide high-quality timber (Figure 3A). However, most *Robinia* forests are in dry habitats such as nutrient-poor sandy or rocky pastures on originally infertile soils threatened by soil erosion (Vadas 1914, Hegi 1924, Göhre 1952, Kolbek et al. 2004), where trees hardly reach 10 m and are often used for firewood or making poles (Vítková and Kolbek 2010, Vítková et al. 2015, 2017). In wooded areas, light-demanding *Robinia* does not spread into dense forests, but is able to colonize forest margins or disturbed sites, such as fresh clear-cuts or post-fire sites. Spontaneously, it spreads also into other open habitats in the vicinity, for example abandoned vineyards, orchards and fields.

Biodiversity value of such *Robinia* forests is mostly low, however, certain groups of organisms prefer them (e.g. macrofungi or habitat generalists among birds; Ślusarczyk 2012, Hanzelka and Reif 2015a, 2015b). The undergrowth is often dense and rich in species (~20 to 45 plant species/200 m^2, similar to that in climax forests), but it is dominated by widely distributed nitrophilous species sharing a wide range of nemoral and ruderal habitats, e.g. *Bromus sterilis*, *Galium aparine*, *Urtica dioica*, *Hedera helix* and *Sambucus nigra.* Species-poor *Robinia* forests growing in dry habitats are dominated by grasses, the dense cover of which may slow down the establishment of native trees.

Establishment and maintenance: Most European production of *Robinia* wood comes from these plantations. In the Pannonian basin in particular they are the main type of forest and their yield varies between 80 and 280 m^3/ha and have an average rotation age of 30 years (Rédei et al. 2008). New stands are still being established, for example in Hungary, Italy and Romania (Rédei et al. 2008, 2012, Enescu and Dănescu 2013, Ciuvăt et al. 2015, Meloni et al. 2016) but not in Czechia, Poland and Switzerland (e.g. MZE 2014, Wojda et al. 2015). Producing saplings from seed is a relatively simple and low cost method, although germination must be facilitated by mechanical scarification (Rédei et al. 2012), soaking in concentrated sulphuric acid or boiling water (Huntley 1990). Propagation from root cuttings is suitable for producing articular clones or special cultivars (Keresztesi 1988, Rédei et al. 2012). Regeneration from root suckers produces a higher yield than from seedlings at a harvest age of 35–37 years. *Robinia* forests need more management than climax tree species (e.g. oaks), as without regular silvicultural treatments the quality of wood deteriorates due to an unshaped crown and deformed trunk (Bělař 2011, Rédei et al. 2012).

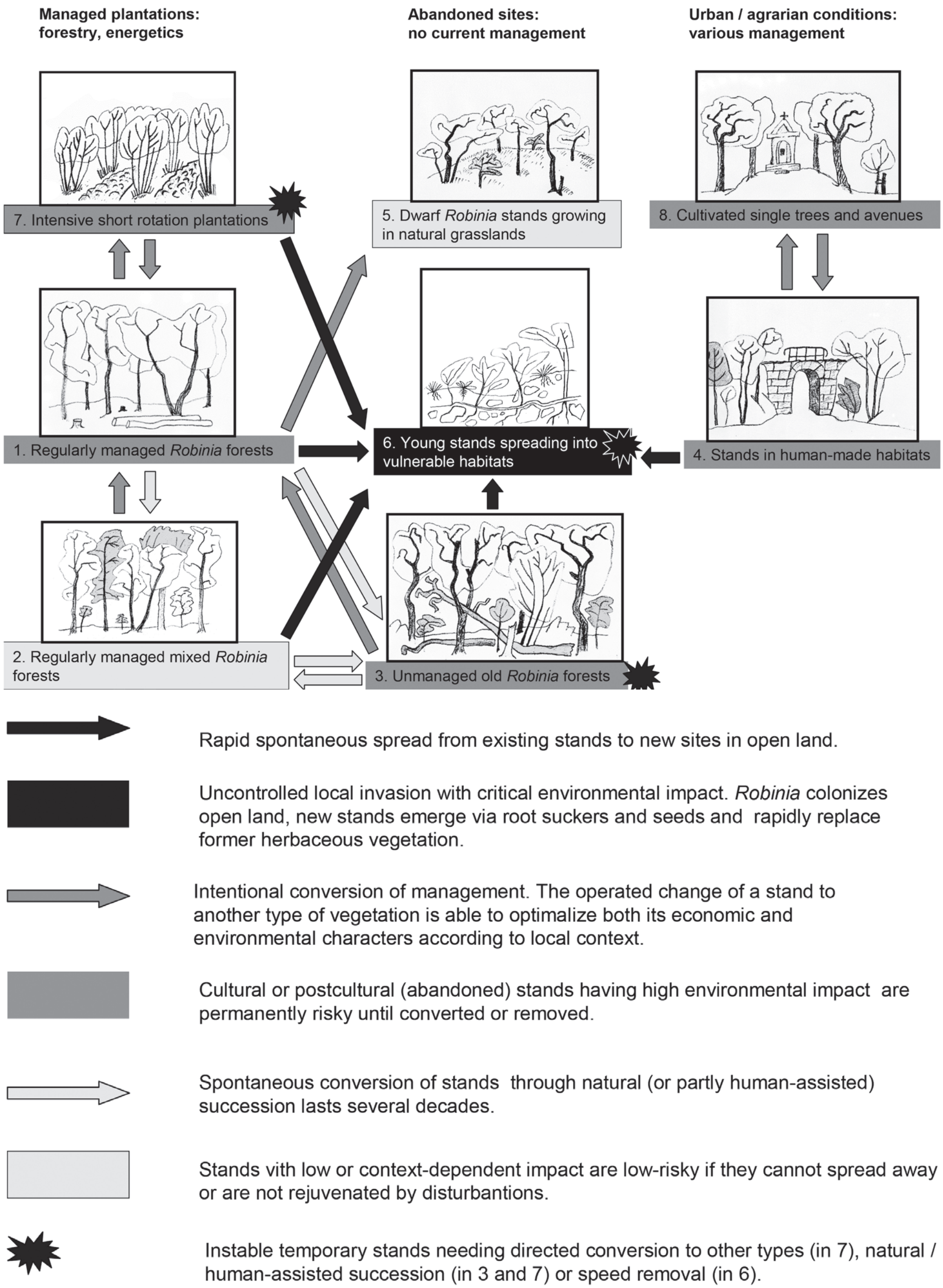

Figure 2. Main successional / intentional dynamic changes among the types of *Robinia* stands. Numbers of vegetation units correspond to stand categorization in the text.

Figure 3. Closed forests in natural habitats (**A–C**) and small-scale stands in man-made habitats (**D**). **A** *Robinia* forest regenerating and managed by coppicing in stripes **B** Planted mixed forest with native *Fraxinus excelsior* and alien *Robinia pseudoacacia* **C** Old *Robinia* forest overgrown by *Fraxinus excelsior* and *Acer platanoides* as a result of spontaneous succession **D** A spontaneously established mixed stand with *Robinia* growing in a quarry.

Tolerance of natural succession: Not remarkable due to economic value of these forests.

Conversion: Restoration of native vegetation is mostly not profitable, being costly and time-consuming. Because of the high sprouting ability of *Robinia*, it is very risky to stop eradication before totally removing all sprouts (Novák 2005, Pergl et al. 2016b, Vítková et al. 2017). There is nothing to be gained by restricting conventional silviculture, especially in early deforested lowlands or suburban zones where *Robinia* has been domesticated for a long time, forms extensive stable metapopulations and where native trees suitable for afforestation are lacking and there are no issues with nature conservation. However, establishment of new stands must be specially assessed, notably those to be established in close proximity of dry or mesic open natural habitats, due to the high sprouting ability of *Robinia*.

2. Regularly managed mixed *Robinia* forests (Table 2, Figures 2, 3B)

Admixture of *Robinia* with cover of up to 10% is the most common type of its occurrence (Vítková et al. 2017). It is a frequent spontaneous admixture in drier parts of hardwood floodplain forests on well-drained and fertile soils, mainly consisting of *Quercus robur*, *Carpinus betulus*, *Ulmus minor* and alien *Ailanthus altissima* (Essl et al. 2003, Terwei et al. 2016). Dry deforested slopes in Czechia were stabilized using *Robinia* and locally alien *Pinus nigra* at the turn of the 19th century.

The environmental impact of *Robinia* growing in mixed stands is considerably less than in monocultures. In closed mature forests, it survives only as individual trees or groups of trees in areas that were previously disturbed. The composition of shaded undergrowth is dependent on the proportion of the canopy that consists of *Robinia* (Essl et al. 2011). Birds benefit from its presence in mixed forests up to approximately equal proportions between *Robinia* and native trees, but its higher share causes shifts in bird community compositions toward a dominance of generalist species at the expense of specialists. This invasive species affects birds by altering structural components of the habitat and related supply of food and cavities for hole-nesting birds (Kroftová and Reif 2017). Mixed *Robinia* forests occur mostly close to native forests and thus *Robinia* does not pose danger for local or surrounding vegetation.

Establishment and maintenance: Reasons for the establishment of these forests were either to supplement natural sparse stands, e.g. forest-steppes with *Quercus* spp. or to improve soil quality, yield and species diversity after logging of native forests and in inter-cropping plantations (Figure 3B; Groninger et al. 1997, Mosquera-Losada et al. 2012). Mixed forests with *Robinia* can be managed as a standard part of current silviculture if some conditions are fulfilled. It is important to reduce light availability inside the forest. Traditional management with regular clear-cuts recurring every 20–30 years creates sunny sites which are suitable for reproduction and vegetative regeneration of *Robinia* and thus drives its invasion into native deciduous forests (Radtke et al. 2013). Such invasion can be accompanied by spread of other weedy or invasive species. Natural disturbances forming light gaps in closed forest canopies, such as trees dying, fire or windthrow are other factors facilitating *Robinia* invasion as the species is highly adapted to disturbance. Under unfavourable light conditions, it develops a persistent bud bank on roots, stems and branches, allowing a rapid reaction to canopy opening following disturbance resulting in the establishment of compact clonal colonies covering areas up to several hundred square meters (Kowarik 1996, Chang et al. 1998, Krízsik and Körmöczi 2000, Schiffleithner and Essl 2016).

Tolerance of natural succession: Natural decline in *Robinia* abundance during succession was observed only in forests without large-scale disturbances, where *Robinia* finally occurs only as an admixture restricted to more open sites (Motta et al. 2009, Somodi et al. 2012, Terwei et al. 2013).

Conversion: Selective cutting that reduces light availability (Radtke et al. 2013) and favours native tree species is needed in such mixed forests. However, efforts to eradicate all *Robinia* trees would be fruitless because of economic demands and risk of failure.

3. Unmanaged old *Robinia* forests (Table 2, Figures 2, 3C)

Protective monodominant forests 12–16 m tall and over 50 years old on steep slopes pose a big problem in terms of their stability. Trees gradually die, are prone to windthrow and damage and the forest becomes more open. The shrub layer is rich in species. The herb layer consists of dominating grasses, relicts or pioneers of natural forest communities and nitrophytes with cover depending on water regime of topsoil. Such protective forests provide excellent honey (Vítková et al. 2017).

Establishment: In some countries (e.g. Czechia and Switzerland), this species was used to stabilize deforested steep eroded hillsides along rivers that were threatened by soil erosion (former pastures) and transport corridors (Vítková et al. 2017). Because of inaccessible terrain, old *Robinia* plantations have remained without management for several decades.

Maintenance: Maintenance or restoration with native species is mostly not profitable. Old trees are often unstable, therefore logging is difficult and risky, and profit is low. Moreover, logging may trigger soil erosion and regeneration of *Robinia*.

Tolerance of natural succession: During spontaneous succession, *Robinia* is replaced by shade-tolerant competitive trees such as *Fraxinus excelsior*, *Acer pseudoplatanus*, *A. platanoides*, *A. campestre* (Figure 3C) or tall shrubs such as *Crataegus monogyna* on dry sites (Vítková 2014). The rate of this succession is greater if populations of native competitors already occur in the understory or in the neighbourhood. Under closed canopies of native species, *Robinia* does not sprout spontaneously or only slightly (Vítková et al. 2016).

Conversion: Slow conversion to natural forest by means of natural succession is recommended, if there is no risk to biodiversity (adjacent natural habitats) or human infrastructure (traffic corridors or built-up sites; Pergl et al. 2016b). To prevent recovery of *Robinia,* it is important to avoid all interventions that induce sprouting, even leaving dead wood at a site after disturbance (e.g. wind break). If necessary to proceed faster, the natural step-wise canopy opening can be supported by killing of vital trees using combination of cutting and incomplete girdling deep into the phloem followed by application of herbicides (Böcker and Dirk 2007). Very slow decay of felled *Robinia* trunks (Schwarze 2007) may be utilized to stabilize slopes. However this is costly and time-consuming and should be used only when other methods or natural succession fail.

4. Stands in human-made habitats (Table 2, Figures 2, 3D)

A common feature of this rather heterogeneous type is a ruderal environment in urban, agrarian, industrial or mining areas (Figure 3D), and a high proportion of aliens including cultivated ornamental woody species in the canopy or understory. The stands are widespread across Europe and differ in their origin (spontaneous vs. planted), structure (forest vs. shrubs or semi-open stands) and composition (pure or mixed stands with different types of undergrowth). Most stands are young with either prevailing isolated tree clumps or strips growing in the peripheries of towns and agrarian landscapes or larger disconnected groves in reclaimed mining areas.

Establishment and tolerance of natural succession: As early as in the 1970's, *Robinia* was used for the biological recultivation of the post-mining landscapes and landfills (e.g. Bellon et al. 1977) as it is still used in many countries in Europe, South Korea and China (Kim and Lee 2005, Grünewald et al. 2009, Wang et al. 2012, Wojda et al. 2015). In mining areas e.g. in Poland, Germany and Czechia, *Robinia* forms planted or spontaneous stands with native pioneer species such as *Betula pendula*, *Pinus sylvestris*, or alien *Populus* hybrids. In urban areas, *Robinia* is at first cultivated, often escapes and overgrows wasteland and public greenery. These *Robinia* stands are accompanied by native nitrophilous trees such as *Acer platanoides* and *Fraxinus excelsior,* and many aliens such as *Prunus cerasifera, Lycium barbarum* and *Parthenocissus quinquefolia*. In agricultural Pannonian lowland, spontaneous and planted *Robinia* stands along roads are commonly admixed with thermophilous alien trees such as *Ailanthus altissima*, *Gleditsia triacanthos*, *Celtis occidentalis* and *Morus alba*.

Maintenance and conversion: Active management is needed since rapid spontaneous changes tend to occur in this habitat. Consideration of the local context (e.g. role of surroundings, ornamental or utility value, claims of owner or public) is necessary, especially in urban areas. Therefore, different parts of the same stand may be managed differently, including e.g., removal of *Robinia* or whole stands. However, there is no reason for eradicating or banning the planting of *Robinia* in urban areas (Sjöman et al. 2016). Some stands with alien species can even be developed within a novel system of urban nature (e.g. in Berlin; Kowarik and Langer 2005). Planting *Robinia* in mining areas does not pose a problem providing its dispersal does not threaten surrounding valuable habitats. Its gradual decrease during natural succession or mechanical control followed by conversion of stands to vegetation with native species is recommended.

5. Dwarf *Robinia* stands growing in natural grasslands (Table 2, Figures 2, 4A)

Most of these stands originated from unsuccessful planting combined with spontaneous spread in dry habitats. *Robinia* survives in very dry habitats where it occurs as small and twisted trees (~5–10 m in height) or even shrubs forming sparse semi-open stands with an admixture of native xerophilous shrubs, e.g. *Crataegus* spp., *Prunus spinosa* and *Rosa* spp. This type is common in the Pannonian lowland (Hungary and adjacent parts of Austria, Czechia, Slovakia and Slovenia) and in Southern and South-eastern Europe.

Establishment and tolerance of natural succession: In some European countries (e.g. Slovakia, Slovenia, Italy), there is a long historical tradition in *Robinia* planting for vineyard poles and wine barrels (at least since the late 19th century; Vítková et al. 2017). Such plantations have been established at sunny and dry sites of low quality – often low stony knolls surrounded by farmland, where ploughing of fields or mowing of meadows have prevented the vegetative spread and survival of *Robinia* seedlings and sprouts (Figure 4A). Slow growth and propagation of *Robinia* together with weak nitrification and low shading effect ensure the survival of these stands and of some plants, fungi, invertebrates and birds of sunny habitats (Vítková and Kolbek 2010, Ślusarczyk 2012, Hanzelka and Reif 2015b). Such stands form stable patches increasing the local biodiversity of deforested land; with some of them having over 60 species/200m^2. Some rare plant species are specifically linked to these stands, such as perennial grasses (*Melica ciliata*, *M. transsilvanica*), geophytes (*Anthericum liliago, Ranunculus illyricus,* many species of *Allium, Gagea, Muscari*, and *Ornithogallum* genera) and xerophilous herbaceous plants (*Hesperis tristis, Verbascum phoeniceum*). Despite high levels of potential nitrification, nitrophytes typical of *Robinia* stands occur only rare, probably due to drought (Vítková et al. 2015).

Maintenance and conversion: It should be left to the nature conservationists to decide whether to tolerate or remove these stands. However, most of these stands are very old and unlike those in mesic habitats, their shrubby growth does not indicate they are young plants with a potential for future growth, but are usually full-grown with their propagation greatly constrained by stress (Vítková et al. 2017). As in previous units, eradication of *Robinia* and restoration of native vegetation would be expensive and very risky. Monitoring succession and restricting spread into surrounding habitats, possibly combined with grazing or mowing seems to be the optimal management strategy.

6. Young *Robinia* stands spreading into vulnerable habitats (Table 2, Figures 2, 4B)

This type, which complements the previous one, refers to current invasion of natural habitats by *Robinia* (Figure 4B). Spontaneous occurrence of the young stages of *Robinia* poses serious threat to the conservation of dry to mesic grasslands and open dry forests as they are the habitats most endangered by this species invasion (Vítková et al. 2017).

Establishment and tolerance of natural succession: Compared to native trees, *Robinia* has a high sprouting ability and is extremely resistant to disturbance. It produces

Figure 4. Non-forest habitats (**A–C**) and *Robinia* in urban environment (**D**). **A** Agrarian landscape with small-scale and semi-open *Robinia* stands. The spread of this species is suppressed by regular use of farming practices **B** Root suckers of *Robinia* invading a thermophilous grassland, which is the habitat of protected plant species **C** Intensive short rotation plantation regenerated by coppicing **D** Avenue of flowering *Robinia* in Prague (Czech Republic).

numerous root suckers that enable it to disperse at up to 1 m per year (Central Europe; Kowarik 1996) or 2 m per year (South Europe; Crosti et al. 2016) in non-forest ecosystems. Especially after disturbance of a tree its roots produce sprouts that grow up to 4m in height per year. On shading by *Robinia* the light regime, microclimate and soil conditions change and endangered light-demanding plants and invertebrates disappear (e.g. Kowarik 1994, Greimler and Tremetsberger 2001, Matus et al. 2003, Vítková and Kolbek 2010). Based on above mentioned reasons, it is not possible to tolerate establishment of *Robinia* plantations and their natural succession on vulnerable habitats, especially dry to mesic grasslands (including sandy steppes and rocky outcrops) and open dry forests as well as areas within a radius of 500 m from them (consistently with http://neobiota.bfn.de).

Maintenance and conversion: The spread should be restricted if *Robinia* stands occur in or adjacent to fallow land, grassland or other habitats with rare native plants, such as those on rocky slopes. The eradication should be rapid and persistent although expensive and risky due to use of herbicides and the disturbance causing vigorous regeneration of *Robinia* and erosion resulting in the release of nutrients and growth of weeds. For detailed list of suitable and unsuitable methods see (Silva et al. 2014, Schmiedel et al. 2015, Pergl et al. 2016b). However, no universally efficient and widely acceptable method seems to exist, because the stem- and root-sprouting ability of *Robinia* is affected by the eradication method as well as by local site conditions. Application of herbicides is necessary, otherwise resprouting of *Robinia* overcomes the effect of grazing or mowing and suckers appear even 30 years after the felling of *Robinia* (Trylč 2007).

Whole *Robinia* clones must be removed as the roots of the individual plants are connected. For quick eradication the best choice is felling followed immediately by spraying the area felled with herbicide. Removal by incomplete girdling (Böcker and Dirk 2007, Schiffleithner and Essl 2016), though demanding and time-consuming, is suitable for inaccessible sites. It is more efficient if combined with herbicide application at the end of summer, when assimilates are translocated to the roots. Elimination of new suckers and seedlings is necessary for at least 3–5 years. Well-proven is long-term grazing by goats once or twice a year, which also prevents the spread of tall weedy grasses. It is also best to remove all the *Robinia* biomass in order to prevent its sprouting and nutrient release. Due to the high dispersal rate of *Robinia*, control should also concentrate on populations adjacent to valuable habitats, at least to the distance of 500 m (consistently with http://neobiota.bfn.de).

7. Intensive short rotation plantations (Table 2, Figures 2, 4C)

Planting short-lived *Robinia* plantations for renewable bioenergy production (Figure 4C) is currently fashionable. Short-lived *Robinia* plantations occur in many countries worldwide, such as Albania, Austria, China, Italy, Germany, Greece, Hungary, Poland, Slovakia, Spain, South Korea and the United States (e.g. Grünewald et al. 2009, Rédei and Veperdi 2009, Stolarski et al. 2013, Zhang 2014, Straker et al. 2015). Other forms of utilization are rare, for example forage (Papanastasis et al. 1998). Energy production is profitable due to its high, early and easily produced dense, fast drying and combustible wood (Rédei et al. 2008). In the reclamation of heaps of industrial waste in post-mining landscapes one can add other benefits of using *Robinia,* such as high drought tolerance and ability to fix nitrogen (Grünewald et al. 2009).

Establishment and maintenance: These plantations should be established only in areas where an abundant metapopulation of *Robinia* already exists. The most common methods are either planting seedlings or rooted cuttings, however, a more environmental friendly and cheaper method is to transform *Robinia* forests at low quality sites (Rédei and Veperdi 2009). Because of its short coppicing period (average 4–5 years), *Robinia* grows to 5–6m in height (Rédei et al. 2010), nutrients in topsoil are depleted (Vasilopoulos et al. 2007) and undergrowth is species-poor and dominated by undemanding weeds. It is important to prevent further spread of *Robinia* (Crosti et al. 2016). Although closed forests are invasion-resistant, the establishment of new plantations in open land, especially at abandoned sites, close to roads or navigable rivers, is not recommended. As a barrier against *Robinia* invasion buffer zones of non-invasive plants (e.g. vineyards, orchards or fields) can be used, because periodic ploughing or harrowing suppress both the vegetative and generative reproduction of *Robinia* (Crosti et al. 2016).

Tolerance of natural succession and conversion: Extreme caution should be taken when such plantations are abandoned. There is a great risk of an intensive growth of suckers of *Robinia*, especially as the spontaneous succession of native vegetation is very slow. In northeastern Greece, succession to near natural riparian forest was not recorded even 14 years after abandonment. Site preparation for establishment of plantations as well as relatively low production of litter and periodic removal of organic matter through wood cutting caused a long-term changes in availability of soil nutrients and light, thereby affected species composition in behalf of ruderal species (Vasilopoulos et al. 2007). Another limitation often is a low pool of native trees in the vicinity and lack of serious natural enemies (Vítková et al. 2017). For successful conversion it is important to eliminate competition from *Robinia* and assist with reforestation using native tree species.

8. Cultivated single trees and avenues (Table 2, Figures 2, 4D)

This type includes individual *Robinia* trees occurring solitarily or in groups in parks, gardens and at sites such as chapels or crossroads (Pergl et al. 2016d), furthermore in lines along roads, streets and rivers, in windbreaks, vineyard boundaries, hedgerows, gullies etc. (Figure 4D). Their function is mainly ornamental, together with protection against dust, noise or wind. Such structures are currently used to protect crops and livestock against weather extremes, for example in Hungary (Takács and Frank 2009). In Germany, "open orchards" consist of belts of vegetables or cereal fields separated by lines of fast-growing trees including *Robinia*, which are coppiced for biomass production and also used for improvement of soil quality and biodiversity (Mosquera-Losada et al. 2012, Medinski et al. 2014). As *Robinia* is a favourite horticultural tree, there are many interesting cultivars that are generally less invasive than the typical form (Hillier and Lancaster 2014).

Establishment, maintenance, tolerance of natural succession and conversion: Planting is usually easy. Trees need to be pruned and suckers removed regularly to prevent invasion into surrounding habitats. Consideration of the local context is necessary, *Robinia* should not be planted close to vulnerable natural habitats. Old trees are desirable because they provide shade and habitat for, e.g., rare saprophytic fungi or saprophagous beetles (Ślusarczyk 2012, Stejskal and Vávra 2013).

Conclusions

Based on the environmental conditions and human land use we reconcile the main contradictory approaches to *Robinia pseudoacacia* in Europe, where it is planted for multiple beneficial purposes, but also escapes from cultivation and becomes invasive, with impact on species diversity and ecosystem functioning. At the moment the management of *Robinia* stands varies locally, depending on the socio-economic benefits vs. biodiversity impacts, from enthusiastic embrace to planting restrictions to complete rejection. Unfortunately, the information sources related to possible management are biased by narrow focus of the parties involved (environmental vs. forestry). Furthermore, the legislation in several European countries governing the management of *Robinia* is often contradictory.

For these reasons, an integrated solution to harmonize the different views of various target groups is needed. We propose a stratified approach to the *Robinia* management, which takes into consideration both the ecological and economic aspects associated with its occurrence. Because *Robinia* grows in a wide range of habitats ranging from urban environment and agricultural landscape, to forest and natural grassland, neither unrestricted cultivation nor large-scale eradication is feasible. We offer several decision scenarios suitable for specific situations in particular landscapes, where *Robinia* is tolerated in selected areas, but eradicated in others. We distinguish eight types of *Robinia* stands; for each of them we describe ecological conditions, economic benefits, and environmental risks and propose sustainable management practices.

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An assessment of the evolution, costs and effectiveness of alien plant control operations in Kruger National Park, South Africa

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Abstract

Alien plants were first recorded in 1937 in the 2 million ha Kruger National Park (KNP, a savanna protected area in South Africa), and attempts to control them began in the mid-1950s. The invasive alien plant control program expanded substantially in the late 1990s, but its overall efficacy has not been determined. We present an assessment of invasive alien plant control operations over several decades in KNP. We based our assessment on available information from a range of control programs funded from various sources, including national public works programs, KNP operational funds, and foreign donor funds. Over ZAR 350 million (~ US$ 27 million) has been spent on control interventions between 1997 and 2016. We found evidence of good progress with the control of several species, notably *Opuntia stricta*, *Sesbania punicea*, *Lantana camara* and several aquatic weeds, often because of effective biological control. On the other hand, we found that over one third (40%) of the funding was spent on species that have subsequently been recognised as being of lower priority, most of which were alien annual weeds. The allocation of funds to non-priority species was sometimes driven by the need to meet additional objectives (such as employment creation), or by perceptions about relative impact in the absence of documented evidence. We also found that management goals were limited to inputs (funds disbursed, employment created, and area treated) rather than to ecological outcomes, and progress was consequently not adequately monitored. At a species level, four out of 36 species were considered to be under complete control, and a further five were under substantial control. Attempts to control five annual species were all considered to be ineffective.

On the basis of our findings, we recommend that more studies be done to determine impacts associated with individual invasive alien species; that the criteria used to prioritise invasive alien species be documented based on such assessments, so that management can justify a focus on priority species; and that funding be re-directed to those species that clearly pose greater threats, and for which other solutions (such as biological control) are not an option.

Keywords

Biological control, invasion, protected area, savanna, Working for Water

Introduction

The mitigation of threats to biodiversity is a principal aim of protected area management worldwide. Large sums of money are spent to address these threats (van Wilgen et al. 2016), which include urban and agricultural encroachment, invasive species and pollution (Salafsky et al. 2008). However, it is also widely acknowledged that funds for conservation are scarce and cannot meet all demands everywhere (Murdoch et al. 2011). If limited funds are to be spent wisely, an initial plan and periodic assessments of management effectiveness are essential (Leverington et al. 2010, Legge 2015, van Wilgen et al. 2016). These assessments are needed to establish whether management interventions are achieving the desired outcomes, and if not, whether or how management could be adapted to become more effective (Foxcroft and Freitag-Ronaldson 2007). Evaluations are also needed to establish whether the outcomes of management are meeting the expectations of long-term investment of public and private resources (Legge 2015).

Millions of dollars have been devoted to the management of invasive alien plants across the globe, including in protected areas, and robust assessments are needed to establish whether the objectives of management are being met. Several accounts of the ecology of alien plant invasions and the philosophy and history of their management in protected areas have been published (e.g. McKinney 2002, Pauchard et al. 2004, Foxcroft and Freitag-Ronaldson 2007, Foxcroft and Downey 2008, Foxcroft et al. 2013). However, there have only been a few quantitative accounts of the costs and effectiveness of management interventions (see McConnachie et al. 2012, Fill et al. 2016, van Wilgen et al. 2016 for some examples). This is often because researchers and managers operate in different environments, with different goals, different performance measures, and different funding streams. This makes large-scale assessments difficult, because available information from one environment is often not adequate for, or relevant to the other. The lack of invasive alien control program assessments is typical of many, if not most, protected areas globally (Naidoo et al. 2006, Wilson et al. 2007). In a review of invasive alien plant control research, Kettenring and Adams (2011) found that very few studies had evaluated the costs of invasive species control, and these authors urged researchers to provide more complete evaluations of the costs and effectiveness of control interventions.

The benefit of assessments lies primarily in their utility for informing the optimization of control approaches and procedures. Thus, assessments should evaluate not only the cost-effectiveness of programs in terms of money spent on alien species and the ecological outcomes, but also those aspects of the program goals, planning and implementation processes that influenced where and how money was allocated. Assessments should also note which species were prioritized for control, why they were targeted, whether management goals are being met and the constraints that may be limiting current approaches. Budget constraints that influence the choice of control options should also be noted, and the management goals which should guide control programs should be interpreted (Epanchin-Niell and Hastings 2010). Despite decades of expenditure in some countries, assessments have largely been limited to documenting annual control costs (e.g. Pimentel et al. 2005, Sinden et al. 2005); reviews of specific approaches (e.g. biological control, Palmer et al. 2010); or had a focus on specific species (e.g. Bonesi and Palazon 2007, Hazelton et al. 2014, Lindenmayer et al. 2015, Dew et al. 2017). For example, Thorp and Lynch (2000) describe how the weeds of national significance were determined for Australia's control program, and Nel et al. (2004) describe species prioritization for South Africa's Working for Water program. Such information should be considered when evaluating how money was allocated to the control of particular species. Assessments of conservation programs have demonstrated how explicit consideration of goals and objectives can help recommendations for improving these programs. For instance, Parr et al. (2009) considered the management framework, including goals and objectives, of biodiversity conservation programs in Kakadu National Park (Australia) and Kruger National Park (South Africa), generally. Their approach was instructive in demonstrating how explicit consideration of management provided insight into the current status and outcomes of biodiversity conservation efforts in these parks.

In this paper we assess the evolution, costs and effectiveness of alien plant control operations in Kruger National Park (KNP), South Africa. The KNP provides an example of a concerted effort to control invasive alien plants over a very large area, and over several decades. The objectives of this study were to 1) document the goals of alien plant management and the plans for achieving them; 2) identify the species targeted for control and the historical costs of their management; 3) document and assess the effectiveness of the management interventions in reducing the abundance or spread rates of the species; and 4) make recommendations for improving the control efforts.

Methods

Study area

The KNP (~2 million ha) became a protected area in 1898, and gained national park status in 1926. It is situated in the northeastern corner of South Africa, along the border with Mozambique. The mean annual rainfall varies between 350 mm in the north and 750 mm in the south. The vegetation is a well-wooded savanna, and seven major river systems traverse the park from west to east. The KNP is one of few protected areas in South Africa in which invasive alien species, particularly plants, have been managed for more than fifty years (Foxcroft and Freitag-Ronaldson 2007, Foxcroft et al. 2008, Foxcroft et al. 2013). Early in the park's history, the intentional planting of ornamental plant species in tourist camps and staff village gardens was the primary source of the majority of alien plant species introductions (Foxcroft et al. 2008). Increasing urbanization and development outside of the boundaries of KNP subsequently facilitated further plant invasion, especially along rivers (Foxcroft et al. 2008), so that the riparian zones became the most severely invaded habitats (Foxcroft and Richardson 2003). Non-riparian areas also became invaded by alien plant species that were dispersed by birds and mammals, or by human use of roads, tourist camps, and gardens (Foxcroft and Richardson 2003, Foxcroft et al. 2008). In 1997, invasive alien plant control operations were substantially expanded as a result of inflows of funding that followed the establishment of a democratically-elected government in 1994 (van Wilgen and Wannenburgh 2016).

General approach to this assessment

Our assessment was based on information and data from a range of sources. The control of invasive alien species in KNP has relied on several different funding streams, including KNP's own sources for ecosystem management, government-sponsored public works programmes, and foreign donor funding. Each of these sources differed with regard to the goals to be achieved, the formats for data storage, and the requirements for progress reporting. Information on invasive alien plant control operations in KNP has generally been recorded for areas where the control teams worked, and these records include the species that were subjected to control, and the costs of control. However, the data were not always recorded consistently or clearly. For example, the boundaries of spatial units on which control teams worked were changed over time, or in some instances only a portion of the spatial unit on record was treated. In other cases, teams worked on alien plant control as well as on other activities, and the costs of each activity were not recorded separately. Some interventions were recorded as having targeted a certain species, whereas in reality several species were treated in the same operation. For these reasons it was often necessary to make assumptions about the distribution of costs, or species targeted, and we were consequently only able to make a broad-scale assessment of control interventions and their effectiveness. Where assumptions were made, these are stated in the descriptions of methods below. Nonetheless, we believe that reporting the outcome of this assessment in the scientific literature is warranted, given the scarcity of such accounts and their importance in terms of addressing the gaps between research, implementation and monitoring the efficiency and cost-effectiveness of control.

Planning and monitoring

Planning and monitoring are essential elements of management, and clear goals and regular assessments of outcomes are necessary to guide interventions and to gauge progress. We reviewed the systems of planning, management and the monitoring of outcomes based on KNP's management plans and protocols, and on published sources describing the development of management philosophy and its implementation (see, for example, Biggs and Rogers 2003, Foxcroft 2004, van Wilgen and Biggs 2011).

Control measures and effectiveness of control

The prioritization of invasive alien plant species, and their assignment to management intervention categories, has been a fairly recent development in KNP. The initial priorities were only determined in 2008, using a multi-criteria decision-support method that prioritized invasive alien plant species in South Africa's savanna biome (Forsyth and Le Maitre 2011). The criteria for prioritizing species included their impact on biodiversity, on ecosystem services, their relative ease of control, and dispersal potential. An original list of 136 species was reduced in 2015 to 28 species, and ranked by KNP-based ecologists and managers according to the level of concern to KNP (species were divided into those of higher and lesser concern, with a separate category for new incursions with scattered populations that should be prevented from spreading; Table 1). We used this classification as a basis for examining the allocation of funding to invasive alien plant control projects. We also reviewed the protocols and methods that were used to control invasive alien plant species in KNP over the past two decades. These protocols or measures were of two broad types: species-based control, and area-based control. Control measures that targeted particular species included (1) management of species with scattered populations; (2) integrated control of aquatic weeds (*Eichhornia crassipes*, *Pistia stratiotes* and *Salvinia molesta*); and (3) biological control of selected species. Control measures that targeted particular areas entailed labour-intensive piece work on contract for either (4) perennial plants or (5) annual plants. In our analysis, we considered these five approaches separately (annual and perennial plants were considered separately to be able to illustrate the amounts spent on each category, Table 2). The overall effectiveness of control on individual species was assessed, based on the experience of the authors, as follows: (1) unknown (insufficient information to determine effectiveness at this stage); (2) ineffective (control measures are having no discernible effect on the species concerned); (3) moderately effective (spread rates are slowed, but not reversed); (4) substantial (spread rates are reversed, and populations are decreasing); and (5) complete (the threat of the species has been eliminated, and no further action is required; this would apply, for example, if a species were eradicated, or where effective biological control alone prevents re-establishment and spread).

Table 1. Priorities assigned to invasive alien plant species in the Kruger National Park.

Priority	Description	Management approach
Species of higher concern for which separate, dedicated control plans should be developed	Species identified as of sufficient importance to justify a species-specific management plan	Species-dependent. Plans are in development for *Parthenium hysterophorus* (aligned with the national-level approach to this species, outlined by Terblanche et al. 2016), *Chromolaena odorata*, *Opuntia stricta*, and aquatic weeds.
Species of higher concern targeted for control through ongoing clearing and follow-up treatments	Species that have established significant invasive populations in KNP.	Control normally involves labour-intensive mechanical clearing conducted by teams funded by public works programmes.
Incursions with scattered populations (either new species, or isolated outbreaks of species with established populations elsewhere in KNP)	Species targeted as a result of them exceeding a threshold (being noted as a new occurrence, and hence requiring immediate attention to prevent further spread).	Targeted clearing at sites where the species occurs at low densities. Control normally executed by teams funded by KNP Conservation Management operational funds.
Species of lower concern	Invasive alien plant species not considered to be a priority for management	Species that should not normally be targeted for control unless they co-occur with priority species.

Costs of control

The cost of invasive alien plant control was assessed for the period 1997–2016, as there were no reliable records for prior periods. We obtained the annual total amounts allocated each year to alien plant control in KNP from various funding sources. Alien plant control interventions associated with public works funding were contracted out to teams at an agreed cost based on the area that required control, the species present, and their cover (see Neethling and Shuttleworth 2013). The public works programs had recorded the costs of contracts in a spatial database that covered the period 2002 onwards to present. The records included the species that were treated, the density of the invasions, the cost of the operation, the number of people employed, and whether the intervention was an initial clearing, or a follow-up to remove emergent seedlings or re-sprouts. We extracted the data on annual costs per alien species from this database. We used these data to determine the proportion of total funds spent on each species between 2002 to present. Public works programmes were initiated in 1997, but detailed records of the distribution of funds were only available from 2002 onwards. In order to estimate the expenditure per species for 1997–2001, we assumed that the annual funds for those years were spent on individual species in the same proportion as from 2002. In an attempt to prevent cleared areas from becoming re-invaded from outside of KNP, teams also operated on land beyond the park boundary (Fig. 1). Due to recent budget cuts and an emphasis on neighbouring private land, these operations have been limited to 1.5 km from the park boundary, but up to 10 km for some streams and perennial rivers that flow into the park. We separated the control costs incurred inside and outside of KNP.

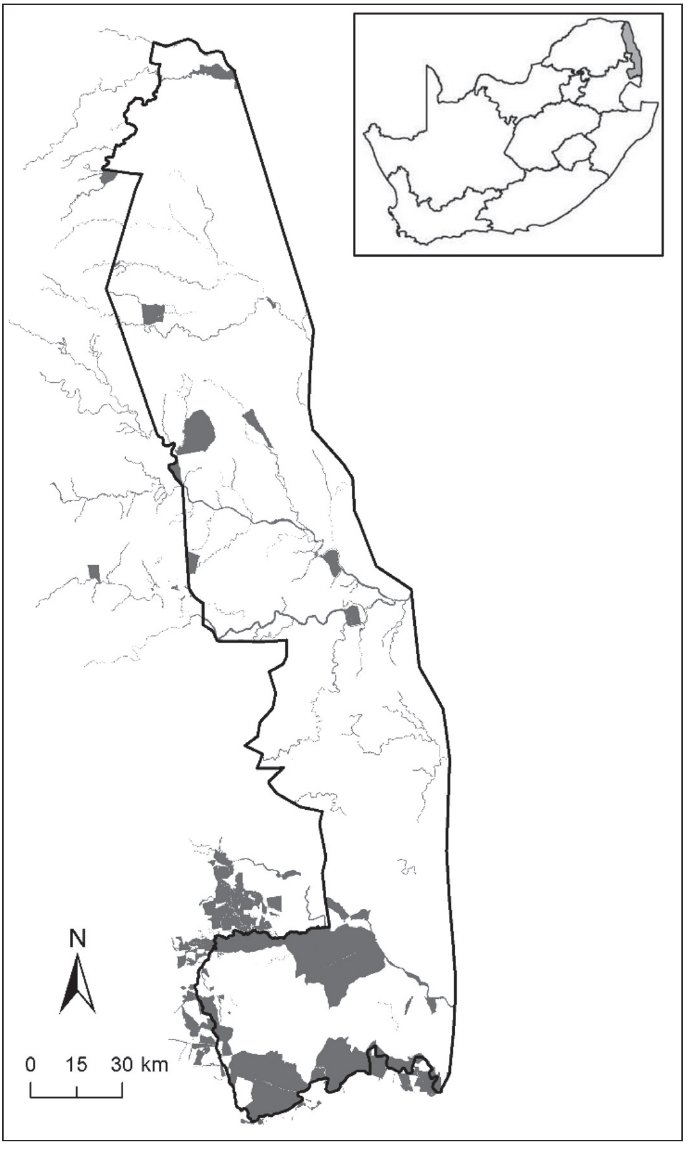

Figure 1. Areas in which alien plant control operations were carried out inside and outside Kruger National Park, South Africa (2002-present). This illustrates the extent of preventative clearing intended to reduce the risk of ongoing invasion from outside of the protected area. The black line delineates the park boundary. Inset shows the location of Kruger National Park within South Africa.

The amounts allocated to alien plant control contracts over the study period accounted for about 60% of the total funds spent. The remaining funds were used for overhead expenses, which included herbicides, training, equipment, supervision, administration and the establishment and operation of mass-rearing facilities for biological control agents. We accounted for overheads by increasing the recorded costs

Table 2. Funding for alien plant management in the Kruger National Park. Funding sources and costs (1997–2016) are associated with five management intervention categories aimed at the control of alien plants in the Kruger National Park.

Management intervention category	Funding source	Description	Duration	Cost (millions of 2016-equivalent ZAR)
Species-based intervention: Management of species with scattered populations	KNP management budget	Mobile team of workers employed by KNP to target isolated populations of invasive alien plants	1982–present	31.4
Species-based intervention: Integrated control of aquatic weeds	Mpumalanga Province	Application of aerial spraying of selected water bodies	2002–present	14.0
Species-based intervention: Biological control of certain species	Public works programs	Targeted programs aimed at the control of selected species	1985–present	Overhead cost, not accounted for in records
Area-based intervention: Labour-intensive piece work to clear perennial invasive alien plants on contract	Public works programs	Contract-based piece work, with the aim of creating employment as well as reducing the spread and extent of invasive alien plants (van Wilgen and Wannenburgh 2016)	1997–present	180.8
Area-based intervention: Labour-intensive clearing by workers employed full-time	Donor funding (Royal Netherlands Government)	Foreign donor funds were used to supplement Public Works funds, with the same goals as for public works programmes	1997–1999	8.3
Area-based intervention: Labour-intensive piece work to clear annual weeds on contract	Public works programs	Contract-based piece work (often with the aim of creating employment).	1997–present	105.6

for each species by a percentage that would bring the total costs for each year up to the full amount spent in that year. To account for inflation, we used the annual consumer price index to inflate all monetary values to 2016 South African Rands (ZAR; 1 US$ ~ ZAR13.5).

Results

Planning and monitoring

The compilation of a management plan is a legislative requirement in South Africa for all protected areas (National Environmental Management: Protected Areas Act, Act 57 of 2003). The KNP management plan (Freitag-Ronaldson and Venter 2009) addressed several themes, one of which was the threats posed by invasive alien species. The KNP's objective with respect to alien species management was "to anticipate, prevent entry and where possible control invasive alien species, in an effort to minimise the impact on, and maintain the integrity of indigenous biodiversity" (Freitag-Ronaldson and Venter 2009: 32). This high-level objective was taken further in separate "management-unit clearing plans" (MUCPs) that provided details of where, and on which species, to focus the funds available for management for a five-year cycle (Foxcroft and McGeoch 2011). At the next level, annual plans of operation were drawn up each year, detailing the allocation of available funds to specific projects.

The KNP has also adopted an overarching philosophy of adaptive management. Under this framework, management interventions are initiated by responding to thresholds of potential concern (Biggs and Rogers 2003). These thresholds are defined for ecosystem indicators, and if a threshold is reached, then management interventions are considered; alternately, the threshold can be recalibrated (Biggs and Rogers 2003). The thresholds for invasive alien species included new occurrences, 5% increases in distribution, and increases in density. In reality, only the first threshold has been used to date due to a lack of data and monitoring (Foxcroft 2009). This system provided further guidance to managers as it identified new priorities for intervention from time to time (see appendix Table 2 in Foxcroft 2009 for examples).

In practice, however, the high-level goal in the KNP management plan has not been effectively carried forward to the 5-yr MUCPs. The MUCPs allocated funding to the control of particular species in particular areas, with goals that quantified the amounts to be spent, the number of people to be employed, and the areas to be treated. Monitoring of outcomes had a focus on these goals, and there were no goals that described the desired outcome in terms of reducing invasive alien plant invasions to manageable levels, what those manageable levels would be, and how long it would take to achieve them (Nicholas Cole, pers. comm.). In the absence of a monitoring program that is focussed on outcomes, it was not possible to objectively assess management effectiveness (see discussion).

Approaches to control

By far the largest proportion of funds was sourced from the nationally-funded public works programs, and was used to fund labour-intensive piece work on contract. The other management intervention categories also made important contributions to the overall outcomes of alien plant management in KNP. These management intervention categories are not entirely mutually exclusive; for example, biological control can make labour-intensive mechanical clearing more effective, if the two are used in tandem. The protocols used in each category are described below.

Species with scattered populations. Once an alien species has invaded an area, targeting isolated or scattered populations delivers the most effective outcomes for containing or reducing the spread of invasions (Higgins et al. 2000). A good example of how this approach has been used in KNP is provided by *Opuntia stricta*, where larger infestations within a defined management area have been managed using the biological control agents *Dactylopius opuntiae* and *Cactoblastis cactorum*, but newly-detected and isolated populations have been targeted for removal using herbicides. In addition, the adaptive management system that identifies alien plant species that have reached a threshold of potential concern constantly generates the need for management capacity to deal with these occurrences as they arise. Management of these instances requires an agile workforce that can be rapidly assigned to new occurrences as they are detected. Such agility is not possible in the case of control projects funded by public works, as contracts are awarded on an annual basis for fixed areas, and cannot be altered. Consequently, this work has been carried out by KNP's own alien biota control team who are permanently employed, and where these constraints do not apply.

Integrated control of aquatic weeds. The management of aquatic invasive alien plants is characterised, in KNP as elsewhere, by a tension between chemical control using aerial spraying and biological control. Chemical control is effective for removing dense invasions on water bodies but needs to be applied repeatedly as surviving plants reinvade the cleared area. In addition, herbicides could have adverse environmental consequences. Biological control, on the other hand, is a more sustainable and benign solution, but it takes longer to become effective, and cannot deal rapidly with large infestations or highly variable seasonal changes (e.g. annual flushing of a river by floods followed by rapid reinvasion). Hill and Coetzee (2017) observed that "while manual removal …. can be successful, it is labour-intensive. Although one of the pillars of the [public works programs] is job creation through alien plant removal, this method is really ineffective for water weeds and this work force [would be] better used on controlling terrestrial weeds in South Africa". Mechanical control of aquatic weeds in KNP is also unacceptably risky due to the presence of hippopotami and crocodiles. Chemical methods have therefore been widely used against aquatic weeds in KNP. *Eichhornia crassipes* was sprayed 2 – 3 times per year on the Letaba and Crocodile Rivers and on some dams, using resources supplied by the Mpumalanga Province. *Pistia stratiotes* and *Salvinia molesta* were additionally targeted with biological control agents, first released in 1985 and 1992, respectively. An example of the tension between chemical and biological control approaches is provided by the case of Sunset Dam, an off-channel water body that is extremely popular with tourists and also heavily invaded by *P. stratiotes* (Fig. 2). Following a decision to stop chemical control of *P. stratiotes* in 1997, the dam became completely covered by *P. stratiotes.* The biological control agent *Neohydronomus affinis* was released in 1997, resulting in the almost total elimination of *P. stratiotes* by October 1998 (MacFadyen et al. 2008). After the initial reduction, the dam reverted to full cover of *P. stratiotes* again by May 1999. This alternating cycle between invaded (complete cover) and clear (complete absence of any plants) persisted for about six years, which was considered unsatisfactory by many managers and tourists. Those responsible for the biological control program were able to resist substantial pressure for the re-introduction of chemical control for long enough, and since May 2004, the dam has remained free of *P. stratiotes* due to the persistence of the biological control agents.

Biological control. Current policy in KNP recognises the imperative to utilize biological control, given that it is relatively cheap, sustainable, and safe (van Driesche and Center 2013, van Wilgen et al. 2013). Biological control in KNP began in 1985, and has been developed in close collaboration with biological control researchers based at the Plant Protection Research Institute and the University of Cape Town. Currently, 22 biological control agents have been released on seven invasive alien plant species in KNP (Foxcroft et al. 2017). Five alien plant species are under either complete control, or the agents contribute substantially to the control thereof (the cactus *Opuntia stricta*, the woody shrub *Sesbania punicea*, and three aquatic species: *Salvinia molesta, Azolla filiculoides* and *Pistia stratiotes*). A facility to breed large numbers of biological control agents has also been established in KNP, with funding from the public works program. This facility supplies biological control agents for distribution across the KNP against several prominent invasive alien plant species (notably the agents for control of *O. stricta*).

Labour-intensive piece work to clear perennial alien plants on contract. This work was conducted by emerging entrepreneurs who were awarded contracts for "piece work". The work itself differentiated between initial clearing or follow-up clearing, to be conducted on a defined area of land and focusing on specific species. Perennial re-sprouting species were typically subjected to an initial clearing in which mature plants were cut at the base and the stumps treated with herbicide to prevent re-sprouting. Treated areas were then revisited on an annual basis to control any re-sprouting stumps with herbicides and to remove or spray emerging seedlings. The total price awarded to each contract was estimated based on the particular species and their density (Neethling and Shuttleworth 2013). The goals of this work were twofold, to control of invasive alien plants and to provide employment. In order to meet the additional goal of maximising employment, and distributing this evenly among communities from all areas adjacent to KNP, projects were distributed across the KNP, several of which may not necessarily have been in areas with concentrations of higher-priority alien plant species. In addition, as found in similar projects, the existence of dual goals resulted in differences of opinion regarding priorities for spending (van Wilgen and Wannenburgh 2016). Cleared areas were frequently revisited to conduct follow-up operations, leading to some concerns among KNP managers that certain areas were being cleared too often (we recorded up to 16 follow-ups on the same site). In addition, annual plans of operation, aligned with MUCP targets, have been inflexible, making it difficult to move the operations to new areas if this became necessary.

Clearing of annual weeds. Annual invasive alien weeds have been extensively targeted in KNP (Table 3). Annual weeds tend to invade disturbed areas in natural ecosystems, especially riparian zones or overgrazed areas (e.g. Morris et al. 2008), where, due to their wide distribution and high abundance in patches, they also provide opportunities to create employment. However, the practice of allocating funds to clearing annual weeds is arguably not always an effective use of scarce resources because annual weeds survive as seeds over the dormant season, and re-appear each year; in addition, most of them (with the notable exception of *Parthenium hysterophorus*) are not known to cause substantial negative impacts; see Discussion).

Effectiveness of control interventions

In the case of KNP, we were not able to systematically assess the effectiveness of control interventions, as these were not effectively monitored. No clear goals were set out in the 5-yr plans (MUCPs), and monitoring was limited to recording the species that were targeted, and the costs of control and follow-up. Nonetheless, there are several approaches that can be used to gauge effectiveness at a broad level. These are discussed briefly below.

Anecdotal evidence of progress: KNP staff and field rangers are generally of the opinion that mechanical and chemical control interventions have been effective in reducing the density of many species, even though there are almost no quantitative data to substantiate this impression. For example, long-serving staff can recall very dense stands of *Lantana camara* along the Sabie River, with impenetrable stands of over 2 m high (K. Maggs, W. Lotter, pers. comm), and these stands are not present today (Fig. 2). Evidence suggests that there was initially a great deal of early effort without demonstrable effect. For example, between 1996 and 1999, KNP teams employed manual labour to remove 8 million stems of *L. camara,* which was widely distributed along rivers in the south of the park (Foxcroft and Freitag-Ronaldson 2007). However, the *L. camara* populations have now apparently been substantially reduced, and this switch is most likely due to an unusually large flood in February 2000 (Heritage et al. 2001) that had a profound influence on the vegetation along the river (Foxcroft et al. 2008). When the floods occurred in 2000, large tracts of riparian vegetation, including almost all infestations of *L. camara*, were swept away (Parsons et al. 2006). This result, combined with intensive post-flood clearing, probably allowed ongoing clearing of *L. camara*, combined with biological control, to become much more effective (Vardien et al. 2012). At the same time, however, the flood disturbance probably facilitated the invasion of other species such as *C. odorata* (Foxcroft and Martin 2002, Leroy 2003). No data existed for the effectiveness of *P. hysterophorus* control either, but this species is spreading rapidly and is recognised as a substantial problem in KNP, as elsewhere (Terblanche et al. 2016). Anecdotal (and photographic) evidence can also be cited in support of progress made with the control of aquatic weeds (Fig. 2).

Assessments of the effectiveness of biological control: The effectiveness of biological control in reducing *O. stricta* invasions is among the most documented of control operations in KNP. Within six years of biological control agents being released in 1988, plant biomass declined by about 90% and has since remained at low levels (Hoffmann et al. 1998, Paterson et al. 2011). No other specific studies of the effectiveness of biological control have been carried out in KNP, but based on assessments elsewhere it appears that the invasive shrub *Sesbania punicea* is under complete biological control in KNP (Hoffmann and Moran 1999). Similarly, biological control has made a substantial contribution to the ongoing management of aquatic weeds, where biological control been demonstrated to have effectively suppressed both *Salvinia molesta* and *Pistia stratiotes* elsewhere in the country (Coetzee et al. 2011).

Short-term studies of effectiveness: In a short-term survey of twelve management units in 2007, Morris et al. (2008) suggested that a single clearing operation reduced

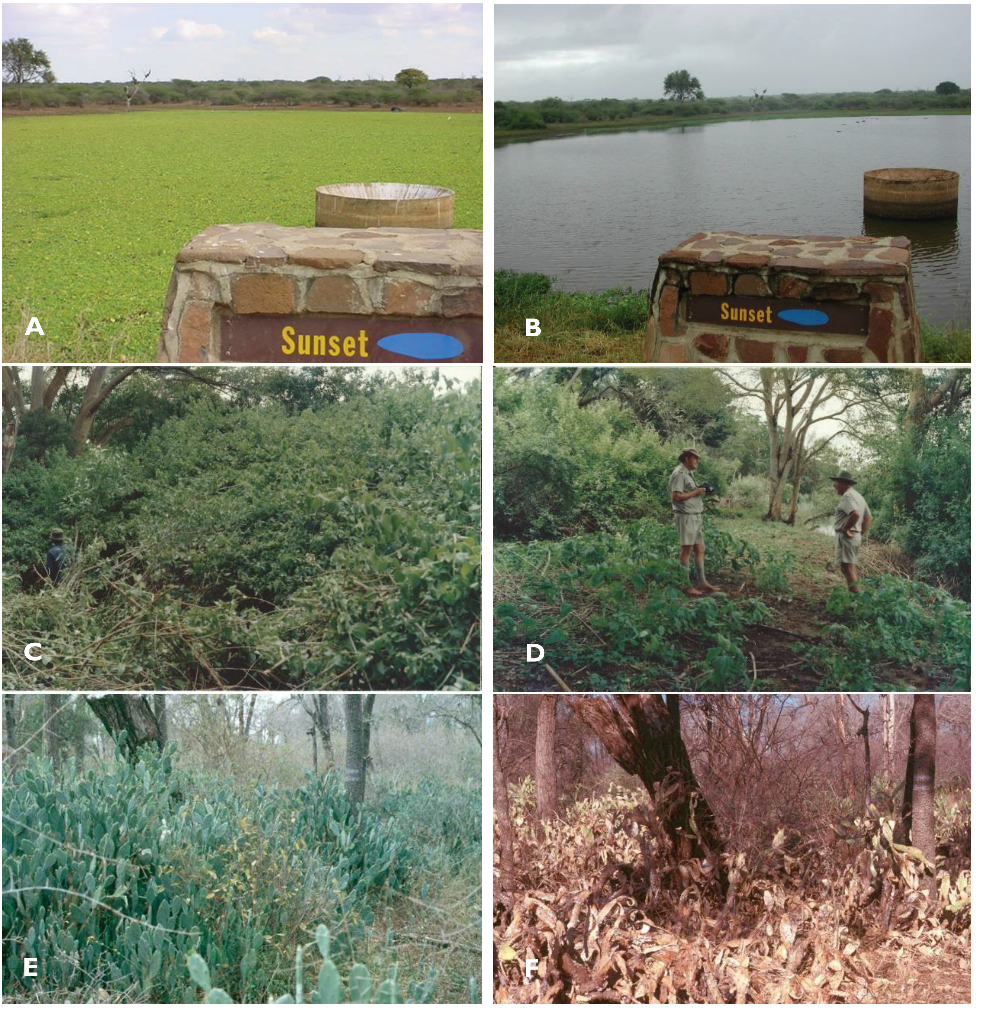

Figure 2. Before and after control of alien plant species in Kruger National Park, South Africa. Sunset Dam was heavily infested by *Pistia stratiotes* (**A**), which was effectively eliminated by a combination of biological and chemical control (**B**). Dense invasions of *Lantana camara* along the Sabie River (**C**) have required intensive mechanical and chemical control to clear (**D**). Populations of *Opuntia stricta* (**E**) have been effectively reduced to low numbers with biological control (**F**).

alien invasive plant densities by 80%. This study concluded that "Continuous clearing acts to effectively limit the establishment and spread of many invasive species despite the ever-present threat of invasion from upstream. Furthermore, the continuous clearing of invasive alien plant stands in KNP ensures that stands are relatively short-lived, preventing long lasting negative impacts on the ecosystem. Removal of invasive alien plant species reduces their disproportionate competitive influence and facilitates the natural re-establishment of native vegetation". This study re-enforces the views of staff above that the densities of some species have decreased.

Genetic studies of source populations: Vardien et al. (2012) used genetics to illustrate that reinvasion of the lower Sabie River in KNP, following the floods of 2000, originated from populations of *Lantana camara* along the tributary Sand River. The Sand River is largely outside of KNP, and was more densely invaded than the Sabie River above the confluence, because of ongoing control on the Sabie that was absent from the Sand River. The study found that re-invasion of the Sabie River below the confluence with the Sand was overwhelmingly from the Sand River populations of *L. camara*. The study concluded that the major flood of 2000 effectively cleared invasive populations of *L. camara* from the riparian areas, and that re-invasion could be attributed to a lack of management outside the KNP, providing evidence of the effectiveness of management in the KNP.

Effectiveness of control of individual species: Based on the experience of the authors, and on the approaches outlined above, it was possible to assign individual species to categories of control effectiveness. Of the 36 species listed in Table 3, four were considered to be under complete control, and a further five were under substantial control. Biological control accounted for all of the species under complete control, and played a role in three of the five species considered to be under substantial control. Control effectiveness was considered to be moderate for two species, and ineffective for five species; control effectiveness for the remaining 16 species could not be assessed with any degree of confidence. Attempts to control annual weeds were all considered to be ineffective.

Costs of control

Over the past 20 years, various organizations have expended almost ZAR350 million (2016 equivalent) on alien plant control operations in KNP (Table 2). Most (84%) of this was funded by public works programs. The largest proportion of public works funding (23%) was spent on the control of *Lantana camara* (Table 3), and most of the funds (61%) were used for clearing outside of the KNP boundary. Just over half (56%) of the funds were expended on species of higher concern, with much less being spent on new incursions with scattered populations (3%; see Table 1 for categories). However, over one third (40%) of the funding was spent on species of lower concern (according to the current classification), of which about half (19% of the total cost of controlling all species) was on annual species of lower concern (Table 3). Because some annual species were regarded as being of higher concern (*Ricinus communis* and *P. hysterophorus*), the amount spent on the control of all annual species was 37% of the total cost. In the case of *Chromolaena odorata*, it is pertinent to note that it was only present as a tiny population in 1997, and it was only once it became more widespread that the spending on this species increased. Had it been present at current densities in 1997, a greater proportion of funding would probably have been directed to its control. The situation is similar for *P. hysterophorus*, although it is a more recent arrival whose spread has been more rapid.

Table 3. Costs and effectiveness of control for selected alien plant taxa in the Kruger National Park. Costs are listed in order of total funds expended per species by public works programs alone between 1997 and 2015 in the Kruger National Park. Cost estimates were not available for aquatic weeds. A proportion of funds were expended outside of the KNP within roughly 10-km of the park boundary. See Table 1 for species priorities, and text for definitions of control effectiveness.

Taxon	**Life form**	**Date first recorded in KNP**	**Priority**	**Cost (millions of 2016-equivalent ZAR)**		**Effectiveness of control**
				Inside KNP	**Outside KNP**	
Lantana camara L.	Perennial shrub	1940	Higher concern (ongoing clearing)	17.4	49.2	Substantial
Ricinus communis L.	Annual shrub	1953	Higher concern (ongoing clearing)	12.8	23.9	Ineffective
Xanthium species, mainly *X. spinosum* L.	Annual forb	1953	Lower concern	15.4	11.6	Ineffective
Senna species, mainly *S. didymobotrya*	Perennial shrub	1952	Lower concern	9.1	11.5	Unknown
Argemone mexicana L.	Annual forb	1932	Lower concern	10.6	7.7	Ineffective
Chromolaena odorata (L.) King & H.E. Robins	Perennial shrub	1997	Higher concern with dedicated control plan	3.6	8.2	Moderate
Datura species	Annual herbs	1953	Lower concern	6.4	5.4	Ineffective
Parthenium hysterophorus L.	Annual herb	2003	Higher concern with dedicated control plan	8.3	3.5	Ineffective
Cardiospermum species, mainly *C. grandiflorum* Sw.	Variable vine	1995	Higher concern (ongoing clearing)	5.3	4.9	Unknown
All other non-priority species	Variable	Variable	Lower concern	3.0	6.3	-
Agave sisalana Perrine	Perennial succulent shrub	1965	Lower concern	0.02	8.1	Substantial
Nicotiana species, mainly *N. glauca* Graham	Perennial shrub	1958	Higher concern (ongoing clearing)	4.9	3.2	Unknown
Solanum seaforthianum Andrews	Perennial vine	1991	Species with scattered populations	2.6	4.5	Unknown
Tecoma stans (L.) Juss. ex Kunth	Perennial shrub or small tree	1950	Lower concern	0.04	5.9	Unknown

Taxon	Life form	Date first recorded in KNP	Priority	Cost (millions of 2016-equivalent ZAR)		Effectiveness of control
				Inside KNP	Outside KNP	
Tithonia diversifolia (Hemsl.)A. Gray	Variable herb or shrub	1953	Higher concern (ongoing clearing)	1.5	3.9	Unknown
Opuntia species (other than *O. stricta*)	Perennial succulent shrubs	1950s	Lower concern	3.8	1.3	-
Melia azedarach L.	Perennial tree	1948	Higher concern (ongoing clearing)	1.0	3.9	Substantial
Psidium guajava L.	Perennial shrub or small tree	1949	Lower concern	0.4	4.2	Unknown
Sesbania species, mainly *S. bispinosa* (Jacq.) W.F. Wight	Perennial tree	1984	Lower concern	1.7	2.3	Complete for *Sesbania punicea* (Cav.) Benth. (biological control); unknown for *S. bispinosa*
Solanum mauritianum Scop.	Perennial shrub or small tree	1954	Higher concern (ongoing clearing)	0.5	3.4	Unknown
Opuntia stricta (Haw.) Haw	Perennial succulent shrub	1953	Higher concern with dedicated control plan	1.1	0.4	Complete (biological control agents redistributed when necessary)
Arundo donax L.	Perennial tall grass	1953	Species with scattered populations	0.3	1.0	Unknown
Agave americana L.	Perennial succulent shrub	?	Higher concern (ongoing clearing)	0.02	0.8	Unknown
Cereus jamacaru DC.	Perennial succulent shrub	1988	Species with scattered populations	0.08	0.4	Complete (biological control)
Dolichandra unguis-cati (L.) L.G. Lohmann	Perennial vine	1965	Species with scattered populations	0.2	0.08	Unknown
Leucaena leucocephala (Lam.) de Wit	Perennial tree	1995	Species with scattered populations	0	0.09	Unknown
Thevetia peruviana (Pers.)	Perennial shrub or tree	1950	Higher concern (ongoing clearing)	0.05	0.03	Unknown
Nerium oleander L.	Perennial shrub	1988	Higher concern (ongoing clearing)	0.03	0.04	Unknown

Taxon	Life form	Date first recorded in KNP	Priority	Cost (millions of 2016-equivalent ZAR)		Effectiveness of control
				Inside KNP	Outside KNP	
Passiflora species, mainly *P. edulis* Sims	Perennial vine	2003	Species with scattered populations	0.0	0.05	Unknown
Cylindropuntia imbricata (Haw.) F.M. Knuth	Perennial succulent shrub	1996	Higher concern with dedicated control plan	0.01	0.01	Substantial (biological control)
Bryophyllum delagoense (Eckl. & Zeyh.) Druce	Perennial succulent shrub	1988	Species with scattered populations	<0.01	<0.01	Unknown
Eichhornia crassipes (C.Mart.) Solms	Aquatic weed	1977	Higher concern	Not available	Not available	Moderate
Pistia stratiotes L.	Aquatic weed	1977	Higher concern	Not available	Not available	Substantial (biological control)
Salvinia molesta D.S. Mitch.	Aquatic weed	1974	Higher concern	Not available	Not available	Complete (biological control agents redistributed when necessary)
Azolla filiculoides Lam.	Aquatic weed	1977	Lower concern	Not available	Not available	Complete (biological control)
Total				110.7	175.8	

Discussion

Current situation

Invasive alien species are regarded as one of the most significant threats to the integrity of KNP (Foxcroft and Freitag-Ronaldson 2007), and this recognition has led in part to the expansion of control programmes (Foxcroft and Freitag-Ronaldson 2007). However, the current KNP management plan (revised in 2008) states that "…alien invasions…are generally currently under reasonable control…" (Foxcroft and Freitag-Ronaldson 2007: 2), and that "The current situation, relating to density and distribution of alien species, is manageable provided careful planning and management remain in place…" (Freitag-Ronaldson and Venter 1996: 54). As outlined above, there is evidence of good progress with the control of several species, notably *Opuntia stricta*, *Sesbania punicea*, *Lantana camara* and several aquatic weeds. The lack of consistent records and monitoring remains a concern, though. As a result there is almost no quantitative evidence that species have been controlled, nor that the measures to control them are appropriate and cost-effective (e.g., Dew et al. 2017), and most assessments (for example those supporting statements in the KNP management plan) come from the undocumented observations of park staff. We would, however, caution against complacency. For example, the relatively recent incursions of the annual weed *P. hysterophorus* into KNP are a cause for serious concern. An isolated recording of the species was first noted in 1991 along the Sand River, and subsequently in a few scattered areas in southern KNP in May 2003. *Parthenium hysterophorus* is an aggressive invader of degraded lands, and it can potentially severely reduce rangeland condition over large areas (Wise et al. 2007). Although there is a dedicated set of protocols for the management of this species in KNP, there has until recently been no monitoring of the effectiveness of management (although this is currently being initiated). As is the case elsewhere in South Africa, the long-term control of this species will probably have to rely heavily on the current efforts to curb further spread and the development of biological control options that will make mechanical and chemical clearing viable (Terblanche et al. 2016).

In addition, although control of invasive alien plants is being achieved within the boundaries for KNP, areas outside of the park remain highly invaded in places (Foxcroft et al. 2007), and thus could continue to act as a source of propagules from which cleared areas in KNP will be re-invaded (e.g. *Lantana camara*, Vardien et al. 2012). Although KNP does operate in a buffer outside of the park, and despite the fact that 61% of available funds were spent outside the park between 2002 and present, the approach faces large challenges, including the need for ongoing negotiation and collaboration between landowners and government agencies. Finally, the expenditure of a large proportion of funds on species of lower concern (especially some annual species) continues to reduce the overall efficiency of the control programme. The focus on annual species has come about for a variety of possible reasons, including the imperative to create employment (annual weeds provide accessible populations for control), the conviction among several managers that they are harmful (but like almost all invasive alien plants, there is no documented evidence of this, see Blackburn et al. 2014), and the fact that annual weeds have not until recently been formally recognised as species of lower concern. In the light of these concerns, we have identified a number of core alien plant control program components that require attention in the interests of improving KNP's invasive alien plant management program, which may provide concepts that can benefit other similar situations.

Planning, goal-setting and monitoring

The practice of setting realistic and achievable goals, based on an agreed set of priorities, the development of plans to achieve these goals, and regular monitoring of outcomes are widely accepted as essential elements of management (Genovesi and Monaco 2013). However, aside from a general goal of maintaining native biodiversity by preventing or controlling alien plant invasions, the KNP's management plans contain no specific measurable objectives or detailed plans for achieving them. The system of using of thresholds of potential concern to guide management interventions is largely aimed at highlighting any changes to a species' situation, and triggering action in response, but it is not designed to guide the management of alien plant invasions that require systematic treatment over multiple years. The practice of allocating available funds to different areas and species without setting clear goals is a widespread shortcoming that has been reported in other parts of the country (Fill et al. 2016; McConnachie et al. 2012, van Wilgen et al. 2012, 2016, Kraaij et al. 2017). The situation could be substantially improved by prioritising the areas to be worked in, setting achievable goals for the control of priority species in priority areas in the MUCPs, practicing conservation triage to ensure that scarce funds are utilised effectively, and expanding the monitoring program to include ecological outcomes in addition to employment creation, disbursement of funds, and areas treated (van Wilgen et al. 2016).

Determining priorities

While KNP has assigned priorities to a number of alien plant species, the allocation of funds to these species did not always reflect these priorities. In particular, a substantial proportion of funding was expended on annual weeds, many of which were later recognised as being of lower priority. Most annual weeds (with the possible exception of *P. hysterophorus*) have not been demonstrated to be harmful, and are only invasive in disturbed areas, including naturally dynamic habitats such as riparian zones or heavily grazed sites. The fact that there are so few studies that document the harmful effects of invasive alien plant species (Jeschke et al. 2013) makes it very difficult to arrive at consensus regarding priorities, and prioritization exercises are consequently influenced predominantly by perceptions. Alien species are regarded as undesirable because they can change biotic interactions and processes in their new range, but many alien species apparently have little or no detectable effects of their new environment (Blackburn et al. 2014). In KNP, the annual shrub *Ricinus communis* is regarded as a priority, even though it never covers large areas at a local scale, i.e. it does not develop into the extensive monocultures associated with other invasive alien species such as *L. camara*, *C. odorata* or *P. hysterophorus* in similar habitats elsewhere in Africa (A.B.R. Witt, Pers. Comm.). Nonetheless, an estimated ZAR 36.7 million has been expended on this species (more than any other species except *Lantana camara*, Table 3), as it is widely perceived as harmful despite a lack of evidence. In addition, given the dual goals of public works projects, funds can be allocated to particular projects to create employment in some areas, rather than to meet ecological goals (van Wilgen and Wannenburgh 2016), leading to further inefficiencies, although we are not able to quantify the degree to which this happens in KNP.

Three responses to this situation seem appropriate. First, it is clear that more studies need to be done to assess the degree of impact associated with individual invasive alien species on which substantial funds are being expended. The resources for conducting these impact assessments should not be sourced from management funds, but rather from the KNP research budget (van Wilgen et al. 2016). Management should be ongoing, but can shift its focus if and when assessments indicate that such a shift would be warranted. Incursions of new alien species can be dealt with without an impact assessment, as control costs would be low, and waiting for a full impact assessment would allow the species to spread, potentially increasing control costs exponentially. Secondly, it would be useful to formally document the criteria used to assign priorities to invasive alien species, so that management can focus on defensible priorities. In this regard, it would be useful to apply the framework developed by Blackburn et al. (2014), which employs the mechanisms of impact used to code species in the International Union for Conservation of Nature (IUCN) Global Invasive Species Database. Finally, although difficult decisions are going to be required, it would seem crucial to re-direct funding to those species that clearly pose greater threats, and for which other solutions (such as biological control) are not an option. Some of these funds could also be used to control alien plant populations outside of KNP, so as to reduce the risk of re-invasion of cleared areas.

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Alien flora of Turkey: checklist, taxonomic composition and ecological attributes

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Abstract
The paper provides an updated checklist of the alien flora of Turkey with information on its structure. The alien flora of Turkey comprises 340 taxa, among which there are 321 angiosperms, 17 gymnosperms and two ferns. Of the total number of taxa, 228 (68%) are naturalized and 112 (32%) are casual. There are 275 neophytes (172 naturalized and 103 casual) and 61 archaeophytes (52 naturalized and 9 casual); four species could not be classified with respect to the residence time. In addition, 47 frequently planted taxa with a potential to escape are also listed. The richest families are *Asteraceae* (38 taxa), *Poaceae* (30), *Fabaceae* (23) and *Solanaceae* (22). As for the naturalized alien plants, the highest species richness is found in *Asteraceae* (31 taxa), *Poaceae* (22), *Amaranthaceae* (18) and *Solanaceae* (15). The majority of alien taxa are perennial (63.8% of the total number of taxa with this life history assigned, including those with multiple life histories), annuals contribute 33.8% and 2.4% are biennial aliens. Among perennials the most common life forms are phanerophytes, of which 20.3% are trees and 12.6% shrubs; woody vines, stem succulents, and aquatic plants are comparatively less represented. Most of the 340 alien taxa introduced to Turkey have their native ranges in Americas (44.7%) and Asia (27.6%). Of other regions, 9.1% originated in Africa, 4.4% in Eurasia, 3.8% in Australia and Oceania and 3.5% in the Mediterranean. The majority of taxa (71.9%) were introduced intentionally, whereas the remaining (28.1%) were introduced

accidentally. Among the taxa introduced intentionally, the vast majority are ornamental plants (55.2%), 10.0% taxa were introduced for forestry and 6.7% as crops. Casual alien plants are most commonly found in urban and ruderal habitats (40.1%) where naturalized taxa are also often recorded (27.3%). Plants that occur as agricultural weeds are typically naturalized rather than casual (16.0% vs 7.1%, respectively). However, (semi)natural habitats in Turkey are often invaded by alien taxa, especially by those that are able to naturalize.

Keywords
Alien flora, Turkey, casual and naturalized alien plants

Introduction

Turkey has a long tradition of floristic research and as a result its native flora is satisfactorily investigated. With more than 12,000 plant taxa (Davis 1965–1985, Davis et al. 1988, Güner et al. 2000, 2012) and new species being continuously described, including new endemics (Güner et al. 2012, Özhatay et al. 2013, 2015), the flora of Turkey is the richest among the Mediterranean, European and neighbouring countries (Ekim and Güner 1986). The majority of this total number is represented by native taxa with 31% of endemics (Güner et al. 2012). Turkey's landscape and ecological diversity has contributed not only to a high floristic richness, but has also allowed for successful introductions and cultivation of a great number of crops, fruit species (Ercisli 2004) and forest trees (Atalay et al. 2014).

On the contrary, up to now there was only limited information on Turkish alien flora. Being located at the crossroads of three continents, there has always been an intense movement of humans and goods across Turkey over the history due to human migration, and in modern Turkey both plants and animals were being introduced intentionally and unintentionally in great quantities. Suitable conditions for the cultivation and use and subsequent naturalization of plants introduced into the country are supported historically. Turkey is a country of special significance in the history of agriculture, with some of the earliest sites of plant domestication nearly 10,000 years ago (Aksoy and Oksar 2015), and today 50% of the country area is agricultural land (FAO 2017).

With this background, it is somewhat surprising that so far, the main source of information about alien flora of Turkey was a checklist generated for the DAISIE project (Delivering Alien Invasive Species Inventories for Europe, 2004–2008; see DAISIE 2008, Lambdon et al. 2008), based on the several decades old flora (Davis 1965–1985) that was rather outdated in terms of inventory of alien species. Therefore, the DAISIE project reported only 220 alien taxa for Turkey, of which only 95 were assigned the naturalization status with certainty (Lambdon et al. 2008), which is an underestimation of the real situation. In fact, it should be taken into account that DAISIE included mainly the European part of Turkey, which represents only 3% of the Turkish territory. More recently, new insights into this aspect were provided by the book “Türkiye İstilâcı Bitkiler Kataloğu” (Catalogue of the invasive plants of Turkey) by Önen (2015).

However, such lack of a recent account on the alien flora represents a serious constraint to the management of those plants that are currently invasive or may become so in the future. As generally agreed, alien species lists form the basis for much of the current research on biological invasions, for guiding legislation and code of conducts, as input to decision making and risk assessment and in the formulation of management policies and strategies for nature conservation (Hoffmann and Broadhurst 2016, Woodford et al. 2016, Jacobs et al. 2017). From the scientific point of view, macroecological analyses of alien floras has received much attention recently and improved the understanding of historical flows of alien species among continents (van Kleunen et al. 2015), the dynamics of their accumulation (Seebens et al. 2017) as well as factors driving the variation in regional diversity of alien floras (Pyšek et al. 2009, 2010, 2015, Essl et al. 2011, Seebens et al. 2015).

The aim of this paper is therefore to fill the important gap in the knowledge on alien flora in one of the richest in species countries in Eurasia, by compiling the first comprehensive list of alien plants in Turkey and providing an analysis of its taxonomic composition, origin and ecological structure.

Methods

Study area

Turkey is a large and diverse country located between 25°40' to 44°48'E, and 35°51' to 42°06'N. The total area is 814,578 km^2 of which 97% is located in Asia and 3% in in Europe. It is divided into seven geographical regions: Black Sea, Eastern Anatolia, South Eastern Anatolia, Mediterranean, Aegean, Marmara and Inner Anatolia. The average altitude is 1,141 m a.s.l., and it increases from West to East; 18% of Turkey is below 500 m and 25% between 500 and 1,000 m. Plains up to 2,000 m of altitude and high plateaus up to 2,500 m are another source of biodiversity of native plants while providing potential diverse niches for the naturalization of alien species. Turkey's natural environment is very diverse in terms of climate, ranging from subtropical to cold temperate, as well as topography and geology (Atalay 2002, 2010, 2011), supporting a variety of vegetation types (Akman and Ketenoğlu 1986). Annual precipitation varies from 300 to 2,000 mm, and mean annual temperature from 4 to 19 °C. Some areas are prone to frosts for almost 10 months, while some have frost for only one day in a year. The growing period varies from almost the whole year to less than 140 growing days. Turkey is surrounded by an 8,333 km coastline with Black Sea at the North, Marmara Sea between two peninsulas, and Aegean Sea at West and Mediterranean at South. The coastal areas represent a dynamic, ecologically fragile environment with threatened habitats in which a diverse range of human activities are carried out (Acar et al. 2014). In addition, the majority of Turkey's ever-increasing population resides in coastal areas (Erginöz and Doğan 1997). Among cities that represent important points of entry of alien species into the country, İstanbul with a population of almost 15 million is Turkey's most populated metropolitan area and the economic powerhouse of the country. Its geographical characteristics and topography allow for the existence of diverse microclimatic zones to exist in a relatively small area of 5,461 km^2 (Güneralp et al. 2013). The 2,875-km long border of Turkey with its neighbours Georgia, Armenia, Azerbaijan, Iran, Iraq, Syria, Greece and Bulgaria is associated with a high probability of entry and occurrence of alien plant species in habitats along adjacent roadside corridors that represent an important pathway for alien plants (Wilson et al. 2016).

Data sources used to compile the inventory

The first flora dedicated to Turkey is composed of the five volumes of Boissier's *Flora Orientalis* (Boissier 1867–1884) and its supplement (Boissier 1888) where alien species are occasionally reported. However, the basic data source used for the present inventory is the Flora of Turkey and the East Aegean Islands (Davis 1965–1985, Davis et al. 1988, Güner et al. 2000, 2012). This source has been complemented with information extracted from all the available literature, such as, in particular, the papers published after 2000 in the Turkish Journal of Botany and elsewhere. In addition, dedicated studies (Uremis et al. 2014, Arslan et al. 2015) and field surveys (e.g. Brundu et al. 2011) were taken into account as well as herbarium samples stored at the Düzce University Forestry Faculty Herbarium (DUOF) and other herbaria in Turkey (GAZİ, ISTO, AİBO and ISTE). We also screened the GBIF database, which holds 265,818 plant records for Turkey (GBIF 2017); however, alien plant species are significantly underrepresented in this source. We also used information from an ongoing project dedicated to the online flora of Turkey (Tübives – http://www.tubives.com/index.php) (Bakis et al. 2011), an initiative for a new Flora of Turkey with illustrations 'Resimli Türkiye Florası Volume 1 (Güner 2014), and 'Bizim Bitkiler' (http://www.bizimbitkiler.org.tr/v2/index.php), another online flora of Turkey which includes the last checklist of vascular flora of Turkey by Güner et al. (2012).

Classification of taxa and their characteristics

This inventory focuses on plant species alien to Turkey (synonyms: exotic, introduced, non-indigenous, non-native), i.e. species present in the country because human actions enabled them to overcome fundamental biogeographical barriers (Richardson et al. 2000, Blackburn et al. 2011); they occur in Turkey as a result of intentional or accidental introduction by humans, or as a result of natural spread from other regions where they were introduced by humans. Crosses resulting from hybridization with one or both alien species involved are also considered alien (Pyšek et al. 2004). In addition, we included in this inventory some taxa that are native to a part of the country but introduced elsewhere in Turkey, i.e. alien in Turkey, following an approach proposed by Lambdon et al. (2008) for Europe.

We classified alien plant species according to the stage they reached along the introduction-naturalization-invasion continuum (Richardson and Pyšek 2006, Richardson et al. 2000, 2011, Blackburn et al. 2011). However, due to a lack of data on the rate of spread we did not classify species as invasive and only classified them in two main categories, casual or naturalized. The complete inventory (Suppl. material 1: Table 1) lists also additional species that are presently recorded only in cultivation outside urban areas, but over very large areas, such as tree species in planted forests, and that could start to naturalize in the future due to potentially strong propagule pressure or climate change. These species are, however, not taken into account for data analyses. Taxa were further classified with respect to their residence time, i.e. separated into archaeophytes and neophytes (see e.g. Pyšek et al. 2004, 2012 for delimitation). Affiliation of taxa to families follows the approach of the Angiosperm Phylogeny Group (Stevens 2001 onwards, APG IV 2016). Plant names have been verified using IPNI (International Plant Name Index, http://www.ipni.org/), The Plant List (2010, version 1, published on the Internet; http://www.theplantlist.org/), WCSP and the African Plants Database (APD, version 3.4.0), updated by the Conservatoire et Jardin botaniques de la Ville de Genève and the South African National Biodiversity Institute, Pretoria, South Africa (http://www.ville-ge.ch/musinfo/bd/cjb/africa). We followed, to our best attempt, the accepted and correct nomenclature according to current taxonomic standards.

Information on life history, region of origin, pathway of introduction (intentional vs accidental) and habitat affiliation was extracted from literature and from the above cited sources for each species.

Life forms were classified as follows: therophytes, hydrophytes, chamaephytes, geophytes, hemicryptophytes and phanerophytes (Raunkiaer 1934, 1937). In addition, growth form and life history were assigned according to the Thesaurus of Plant Characteristics for Ecology and Evolution (Garnier et al. 2017) and other specific literature (Pérez-Harguindeguy et al. 2016). Growth-forms reported for aquatic plants follow Brundu (2015).

The checklist has been archived on the Global Biodiversity Information Facility (Uludag et al. 2017).

Statistical analysis

Differences in representation of life forms within casual and naturalized species were tested by contingency tables with control for overdispersion (if needed using quasi-Poisson distribution) (Crawley 2007). To test individual differences among life forms and species groups, adjusted standardized residuals of G-tests were compared with critical values of a normal distribution (Řehák and Řeháková 1986). All analyses were performed in R 3.0.2 (R Core Team 2015).

Results

Species numbers and taxonomic composition

The alien flora of Turkey comprises 340 taxa, among which there are 321 angiosperms, 17 gymnosperms and two ferns. Of the total number of taxa, 228 (67.1%) are naturalized and 112 (32.9%) are casual (Appendix 1; for the complete list of taxa, which includes additional 47 frequently planted taxa noted above, see Suppl. material 1). Related to the total plant diversity of ~12,000 species in the Turkish flora, the contribution of alien taxa is ~2.8% and that of naturalized taxa ~1.9%. Of the taxa for which the classification according to residence time was possible, there are 275 neophytes (172 naturalized and 103 casual) and 61 archaeophytes (52 naturalized and 9 casual).

Turkey's alien flora includes representatives of 92 families and 251 genera. There are seven families with at least 10 aliens that together comprise 44.7% of the total alien taxa richness of the country; the richest are *Asteraceae* (38 taxa, corresponding to 11.2% of all aliens), *Poaceae* (30, 8.8%), *Fabaceae* (23, 6.8%) and *Solanaceae* (22, 6.5%). As for the naturalized alien plants, the highest species richness is found in *Asteraceae* (31 taxa, 13.6% of the total number of naturalized aliens), *Poaceae* (22, 9.6%), *Amaranthaceae* (18, 7.9%) and *Solanaceae*. Over a half of the naturalized alien richness (51.8%) is concentrated in eight families that contain more than four naturalized taxa (Table 1).

The most represented genus is *Amaranthus* with 13 taxa that are all naturalized, contributing thus 3.3% and 5.7% to all aliens and naturalized aliens, respectively. *Solanum* is also rather rich in aliens, but of the 11 taxa only five are naturalized. Other genera, that are represented by more than five species and the naturalization success of their representatives is high, are *Euphorbia* (88.9% of all aliens in the genus are naturalized), *Acacia* (83.3%) and *Oxalis* (100%). The 11 genera with at least four alien taxa in Turkey together account for 17.6% of the total alien plant richness and 26.3% of the naturalized richness of the country (Table 2).

Ecological attributes

The majority of alien taxa are perennial (63.8% of the total number of taxa with this life history assigned, including those with multiple life histories), annuals are also greatly represented (33.8%) and only 2.4% are biennials. Among perennials the most common life forms are phanerophytes, i.e. trees (20.3%) and shrubs (12.6%); woody vines, stem succulent, bambusoid and aquatic plants are comparatively less represented. There were significant differences in the counts per life history between casuals and naturalized species (χ^2 = 29.85, DF = 0,6, $p<0.001$). This significant difference was due to annuals (therophytes) where the observed counts were higher than expected by chance for naturalized species and lower for casuals and due to woody species (phanerophytes) where the situation was reversed (Figure 1).

Table 1. The most represented families in the alien flora of Turkey, ranked according to the total number of alien taxa, with their representatives classified according to their status. For each family, the number of casual and naturalized taxa and the percentage of naturalized among total aliens are provided. Family names follow APG classification (Stevens 2001 onwards, APG IV 2016).

Family	Total no. of alien taxa	No. of casual taxa	No. of naturalized taxa	% of naturalized taxa
Asteraceae	38	7	31	81.6
Poaceae	30	8	22	73.3
Fabaceae	23	11	12	52.2
Solanaceae	22	7	15	68.2
Amaranthaceae	18	0	18	100.0
Euphorbiaceae	11	1	10	90.9
Rosaceae	10	6	4	40.0
Cupressaceae	9	3	6	66.7
Pinaceae	8	4	4	50.0
Oxalidaceae	7	0	7	100.0
Sapindaceae	7	2	5	71.4
Convolvulaceae	6	2	4	66.7
Aizoaceae	5	0	5	100.0
Apocynaceae	5	2	3	60.0
Moraceae	5	3	2	40.0

Table 2. The most represented genera in the alien flora of Turkey, classified according to their status. For each genus, number of casual and naturalized taxa and percentage of naturalized among total aliens in the genus are provided. Genera are ranked according the total number of alien taxa.

Genus	Total no. of alien taxa	No. of casual taxa	No. of naturalized taxa	% of naturalized taxa
Amaranthus	13	0	13	100.0
Solanum	11	6	5	45.5
Euphorbia	9	1	8	88.9
Oxalis	7	0	7	100.0
Acacia	6	1	5	83.3
Acer	4	1	3	75.0
Bidens	4	0	4	100.0
Cotoneaster	4	1	3	75.0
Erigeron	4	0	4	100.0
Ipomoea	4	0	4	100.0
Paulownia	4	4	0	0.0
Physalis	4	0	4	100.0

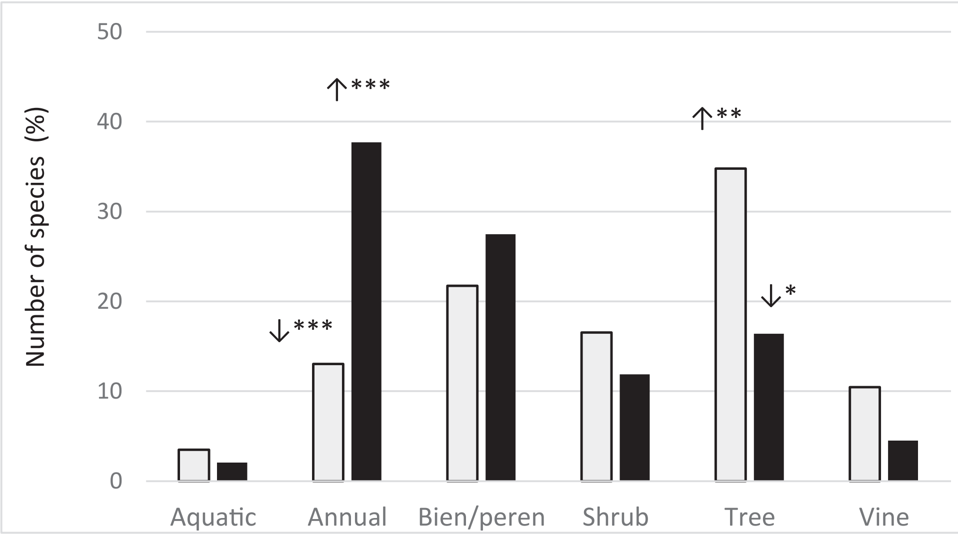

Figure 1. Frequency of alien species in the flora of Turkey categorized according to their Raunkiaer's life forms, shown separately for casuals (white bars, n = 112) and naturalized taxa (black bars, n = 228). Bars indicate the percentage contribution of each life form to the total numbers of incidences within casual and naturalized. Significant differences and their directions are indicated above bars (. < 0.1, * < 0.05, ** < 0.01, *** < 0.001).

Table 3. Structure of the alien flora of Turkey according to origin and number of casual and naturalized species, with percentages of naturalized taxa among total aliens.

Native range	Total no. of alien taxa	No. of casual taxa	No. of naturalized taxa	% of naturalized taxa
America	152	48	104	30.6
Asia	94	33	61	17.9
Africa	31	13	18	5.3
Eurasia	15	2	13	3.8
Australia & Oceania	13	8	5	1.5
Mediterranean	12	1	11	3.2
Europe	9	1	8	2.4
Garden origin & hybrids	8	5	3	0.9
Other & unknown	6	1	5	1.5

Most of the 340 alien taxa introduced to Turkey have their native ranges in Americas (44.7%) and Asia (27.6%). Of other regions, 9.1% originated in Africa, 4.4% in Eurasia, 3.8% in Australia and Oceania, and 3.5% in the Mediterranean (see Table 3 for species numbers with respect to the area of origin).

The majority of taxa in the Turkish alien flora (71.9%) were introduced intentionally, whereas the remaining (28.1%) were introduced accidentally. Among the taxa

Table 4. Habitats in which the alien plant taxa are found in Turkey, shown separately for casual and naturalized taxa, with percentages of the total shown for each category. Natural/semi-natural habitats include the categories of the CORINE Land cover class 3 (Forest and semi-natural areas).

Habitat	Casual alien	%	Naturalized alien	%
Natural/semi-natural habitats	56	28.4	145	28.3
Urban/ruderal habitats	79	40.1	140	27.3
Coastal habitats	34	17.3	96	18.7
Agricultural land	14	7.1	82	16.0
Riparian habitats/wetlands/lakes	14	7.1	50	9.7

introduced intentionally, the vast majority are ornamental plants (55.2%), 10.0% taxa were introduced for forestry (planted forest, reforestation, sand dune stabilization or soil protection) and 6.7% as crops (i.e. plant taxa cultivated for the production of food, forage, fruit, fibre, dye or drugs).

Casual alien plants are most commonly found in urban and ruderal habitats (40.1% of their total number) where naturalized taxa are also often recorded (27.3%). Plants that occur as agricultural weeds are typically naturalized rather than casual (16.0% vs 7.1%, respectively. However, (semi)natural habitats in Turkey are often invaded by alien taxa, especially by those that are able to naturalize (Table 4).

Discussion and conclusions

This is the first comprehensive compilation and analysis of all available records on alien plant taxa in Turkey. It provides the first assessment of their status, introduction purposes and main types of invaded habitats. It also pinpoints knowledge gaps in the geographic and biogeographic distribution and the quantification of environmental and economic impacts.

The total number of the alien taxa reported for Turkey here (340) is relatively low compared to other Mediterranean and Southern European countries, namely France (1,258 taxa), Italy (1,023), Spain (933) and Portugal (547) (Lambdon et al. 2008, Celesti-Grapow et al. 2009) and numerically comparable with Greece (343; Arianoutsou et al. 2010, Dimopoulos et al. 2016). The same is true for the naturalized species richness in Turkey (228 taxa), for which higher numbers are reported for e.g. France (732), Spain (495) or Italy (440), but comparable numbers for Portugal (261) and lower for Greece (134) (Lambdon et al. 2008). This fact, together with the remarkably high richness of native flora of Turkey, makes the contribution of alien species to the total plant diversity of the country relatively low, with the values between 1.9 and 2.8% being by an order of magnitude lower than in some other European countries (e.g. Pyšek et al. 2012) or this continent as a whole. Europe, with a comparable native plant diversity as Turkey, ~10,000 native species (Winter et al. 2009), harbours 1,780 naturalized aliens from overseas and if one considers also intracontinental aliens the number reaches 3,749 taxa (Lambdon et al. 2008) or 4,140 according to the most recent account in GloNAF database (van Kleunen et al. 2015).

This is the first comprehensive catalogue for Turkey and it is based mainly on literature and herbarium data, with only a limited number of dedicated field surveys. Other Mediterranean countries such as France, Italy or Spain have a longer tradition of floristic research on alien plants, whose appearance and establishment have long been documented by botanists there (e.g., by Saccardo 1909). It is therefore possible that casual species are underestimated in the dataset, as casuals in general, and escaped ornamentals in particular (Pergl et al. 2016b), are rarely recorded in botanical works nor are they often collected in herbaria. Another possible explanation for the lower number of alien plants than in some other European countries is that although cultivation of ornamental plants dates back to ancient times, there has been rapid development and change in the ornamental plants sector in Turkey only after the 1980s and this development has gained speed only in the 2000s (Çelik and Arisoy 2013).

The rate of naturalization (proportion of naturalized to all aliens) is 67% in Turkey, i.e. the same as in Cyprus but higher than in Greece (41%), Spain (53%), Portugal (47%) and Italy (51%) (Arianoutsou et al. 2010). On the contrary, with the exception of Bulgaria, there is only very limited knowledge on the alien flora of Georgia, Armenia, Azerbaijan, Iran, Iraq, Syria which impedes comparisons between these countries and, at the same time, forecasting of future trends for the entire Mediterranean region.

National inventories of alien plants are one of the key components for evaluating the status of biodiversity in a given country, as well as threats to endangered species, and provide source data for creating relevant indicators (Lambdon et al. 2008, Celesti-Grapow et al. 2010, Pyšek et al. 2012, van Kleunen et al. 2015, Latombe et al. 2017). Such data are needed for early warning systems, prioritization of management and implementation of effective policy measures (Brunel et al. 2010). The publication of checklists also helps neighbouring countries and trading partners to assess the threat from potential invasions of new species to arrive and checklists can contribute to so-called horizon scanning exercises looking for potential new threats (Roy et al. 2014, Latombe et al. 2017).

Identifying those species that represent potential or future threats, while still at an early stage of invasion, represents a major challenge for prediction (Lambdon et al. 2008, Brunel et al. 2010). Detailed knowledge of the pool of alien naturalized species from which emerging invaders recruit can provide national authorities in Turkey with an instrument for prioritization of management measures and allocation of resources to those species where future spread, and environmental and socioeconomic impacts are likely to occur (Brunel et al. 2010, Pergl et al. 2016a, Rumlerová et al. 2016). The results of the present research will increase the awareness of alien plant taxa in Turkey and neighbouring countries and trigger further dedicated specialized studies, such as assessment of the impact by using standard scoring systems (e.g. Blackburn et al. 2014, Nentwig et al. 2016). New alien species are bound to arrive and spread in Turkey and we hope that publication of this list will encourage further recording so that the impacts of these species can be minimized.

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Appendix 1

Table A1. List of naturalized and casual alien taxa in the flora of Turkey. Taxa are ordered alphabetically. Each taxon is listed together with its family, residence time (Res: Arc = archaeophyte, Neo = neophyte); invasion status (Stat: Cas = casual, Nat = naturalized), simplified growth form and native range.

Taxa	Family	Res	Stat	Simplified growth form	Native range
Abutilon theophrastii Medik.	*Malvaceae*	Arc	Nat	Herb	Asia
Acacia dealbata Link	*Fabaceae*	Neo	Cas	Tree	Australia
Acacia karroo Hayne	*Fabaceae*	Neo	Nat	Tree	Africa
Acacia longifolia (Andrews) Willd.	*Fabaceae*	Neo	Nat	Tree	Australia
Acacia mearnsii De Wild.	*Fabaceae*	Neo	Nat	Tree	Australia
Acacia retinodes Schltdl.	*Fabaceae*	Neo	Nat	Tree	Australia
Acacia saligna (Labill.) H.L.Wendl.	*Fabaceae*	Neo	Nat	Tree	Australia
Acalypha australis L.	*Euphorbiaceae*	Neo	Nat	Herb	Asia
Acer buergerianum Miq.	*Sapindaceae*	Neo	Nat	Tree	Asia
Acer negundo L.	*Sapindaceae*	Neo	Nat	Tree	America
Acer palmatum Thunb.	*Sapindaceae*	Arc	Nat	Tree	Asia
Acer saccharum Marsh.	*Sapindaceae*	Neo	Cas	Tree	America
Acorus calamus L.	*Acoraceae*	Arc	Nat	Aquatic	Asia
Actinidia deliciosa (A.Chev.) C.F.Liang & A.R.Ferguson	*Actinidiaceae*	Neo	Cas	Vine	Asia
Aesculus carnea J.Zeyh.	*Sapindaceae*	Neo	Nat	Tree	Garden/Hybrid
Aesculus hippocastanum L.	*Sapindaceae*	Neo	Nat	Tree	Europe
Agave americana L. var. *americana*	*Asparagaceae*	Neo	Nat	Succulent	America
Agave americana var. *striata* Trel.	*Asparagaceae*	Neo	Nat	Succulent	America
Agrostemma githago L.	*Caryophyllaceae*	Arc	Nat	Herb	Mediterranean
Ailanthus altissima (Mill.) Swingle	*Simaroubaceae*	Neo	Nat	Tree	Asia
Albizia julibrissin Durazz	*Fabaceae*	Neo	Nat	Tree	Asia
Alternanthera sessilis (L.) R.Br. ex DC.	*Amaranthaceae*	Neo	Nat	Herb	Asia
Amaranthus albus L.	*Amaranthaceae*	Arc	Nat	Herb	America
Amaranthus blitoides S.Watson	*Amaranthaceae*	Arc	Nat	Herb	America
Amaranthus blitum L. subsp. *blitum*	*Amaranthaceae*	Arc	Nat	Herb	Eurasia
Amaranthus blitum subsp. *emarginatus* (Salzm. ex Uline & Bray) Carretero, Muñoz Garm. & Pedrol	*Amaranthaceae*	Arc	Nat	Herb	Eurasia
Amaranthus blitum subsp. *oleraceus* (L.) Costea	*Amaranthaceae*	Arc	Nat	Herb	Eurasia
Amaranthus cruentus L.	*Amaranthaceae*	Neo	Nat	Herb	America
Amaranthus deflexus L.	*Amaranthaceae*	Neo	Nat	Herb	America
Amaranthus graecizans L.	*Amaranthaceae*	Neo	Nat	Herb	Mediterranean
Amaranthus hybridus L.	*Amaranthaceae*	Neo	Nat	Herb	America
Amaranthus hypochondriacus L.	*Amaranthaceae*	Neo	Nat	Herb	America
Amaranthus retroflexus L.	*Amaranthaceae*	Neo	Nat	Herb	America
Amaranthus spinosus L.	*Amaranthaceae*	Neo	Nat	Herb	America
Amaranthus viridis L.	*Amaranthaceae*	Neo	Nat	Herb	America
Ambrosia artemisiifolia L.	*Asteraceae*	Neo	Nat	Herb	America
Ambrosia tenuifolia Spreng.	*Asteraceae*	Neo	Nat	Herb	America

Taxa	Family	Res	Stat	Simplified growth form	Native range
Ammannia coccinea Rottb.	*Lythraceae*	Neo	Nat	Herb	America
Amorpha fruticosa L.	*Fabaceae*	Neo	Cas	Shrub	America
Araujia sericifera Brot.	*Apocynaceae*	Neo	Nat	Vine	America
Armeria maritima (Mill.) Willd.	*Plumbaginaceae*	Arc	Cas	Herb	Europe
Artemisia annua L.	*Asteraceae*	Neo	Nat	Herb	Asia
Artemisia verlotiorum Lamotte	*Asteraceae*	Neo	Nat	Herb	Asia
Arundo donax L.	*Poaceae*	Arc	Nat	Bambusoid	Asia
Aster subulatus (Michx.) Hort. ex Michx.	*Asteraceae*	Neo	Nat	Herb	America
Avena byzantina K.Koch	*Poaceae*	Arc	Cas	Herb	Garden/Hybrid
Azolla filiculoides Lam.	*Azollaceae*	Arc	Nat	Aquatic	America
Bauhinia variegata L.	*Fabaceae*	Neo	Nat	Tree	Asia
Berberis veitchii C.K.Schneid.	*Berberidaceae*	Arc	Nat	Shrub	Asia
Berberis thunbergii DC.	*Berberidaceae*	Arc	Nat	Shrub	Asia
Bidens bipinnata L.	*Asteraceae*	Neo	Nat	Herb	Asia
Bidens campylotheca Sch.Bip.	*Asteraceae*	Neo	Nat	Herb	America
Bidens cernua L. s.l.	*Asteraceae*	Neo	Nat	Herb	America
Bidens frondosa L.	*Asteraceae*	Neo	Nat	Herb	America
Bougainvillea buttiana Holttum & Standl.	*Nyctaginaceae*	Neo	Nat	Vine	America
Bougainvillea glabra Choisy	*Nyctaginaceae*	Neo	Cas	Vine	America
Bougainvillea spectabilis Willd.	*Nyctaginaceae*	Neo	Nat	Vine	America
Brachychiton populneus (Schott & Endl.) R.Br.	*Sterculiaceae*	Neo	Nat	Tree	Australia
Bromus tectorum L.	*Poaceae*	N/A	Nat	Herb	Eurasia
Broussonetia papyrifera (L.) L'Hér. ex Vent.	*Moraceae*	Neo	Nat	Tree	Asia
Bryophyllum delagoense (Eckl. & Zeyh.) Druce	*Crassulaceae*	Neo	Cas	Succulent	Africa
Buddleja davidii Franch.	*Scrophulariaceae*	Neo	Nat	Shrub	Asia
Caesalpinia gilliesii (Hook.) D.Dietr.	*Fabaceae*	Neo	Nat	Shrub	America
Calendula officinalis L.	*Asteraceae*	Arc	Nat	Herb	Eurasia
Callistemon citrinus (Curtis) Skeels	*Myrtaceae*	Neo	Cas	Tree	Australia
Callistemon viminalis (Sol. ex Gaertn.) G.Don	*Myrtaceae*	Neo	Cas	Tree	Australia
Camellia japonica L.	*Theaceae*	Arc	Nat	Shrub	Asia
Canna indica L.	*Cannaceae*	Neo	Nat	Bambusoid	America
Caragana arborescens Lam.	*Fabaceae*	Neo	Nat	Shrub/Tree	Asia
Carex vulpinoidea Michx.	*Cyperaceae*	Neo	Nat	Herb	America
Carpobrotus acinaciformis (L.) L.Bolus	*Aizoaceae*	Neo	Nat	Succulent	Africa
Carpobrotus edulis (L.) N.E.Br.	*Aizoaceae*	Neo	Nat	Succulent	Africa
Carthamus tinctorius L.	*Asteraceae*	Arc	Cas	Herb	Asia
Cascabela thevetia (L.) Lippold	*Apocynaceae*	Neo	Cas	Tree	America
Catalpa bignonioides Walter	*Bignoniaceae*	Neo	Nat	Tree	America
Cedrus atlantica (Endl.) Carrière	*Pinaceae*	Neo	Cas	Tree	Africa
Cedrus deodara (Roxb. ex D.Don) G.Don	*Pinaceae*	Neo	Nat	Tree	Asia
Ceiba speciosa (A.St.-Hil.) Ravenna	*Malvaceae*	Neo	Nat	Tree	America
Cenchrus incertus M.A.Curtis	*Poaceae*	Arc	Nat	Herb	America
Centaurea pullata L.	*Asteraceae*	Arc	Nat	Herb	Mediterranean
Chaenomeles japonica (Thunb.) Lindl. ex Spach	*Rosaceae*	Neo	Cas	Shrub	Asia
Chenopodium album L.	*Amaranthaceae*	Arc	Nat	Herb	Eurasia

Taxa	**Family**	**Res**	**Stat**	**Simplified growth form**	**Native range**
Chenopodium giganteum D.Don	*Chenopodiaceae*	Arc	Nat	Herb	Asia
Cichorium endivia L.	*Asteraceae*	Arc	Cas	Herb	Asia
Citrullus colocynthis (L.) Schrad.	*Cucurbitaceae*	Arc	Cas	Vine	Eurasia
Citrus trifoliata L.	*Rutaceae*	Neo	Cas	Tree	Asia
Coix lacryma-jobi L.	*Poaceae*	Neo	Nat	Herb	Asia
Commelina communis L.	*Commelinaceae*	Neo	Nat	Herb	Asia
Convolvulus tricolor L.	*Convolvulaceae*	Arc	Cas	Vine	Mediterranean
Cortaderia selloana (Schult. & Schult.f.) Asch. & Graebn.	*Poaceae*	Neo	Cas	Bambusoid	America
Cosmos bipinnatus Cav.	*Asteraceae*	Neo	Cas	Herb	America
Cotoneaster adpressus Bois	*Rosaceae*	Neo	Cas	Shrub	Asia
Cotoneaster franchetii Bois	*Rosaceae*	Neo	Nat	Shrub	Asia
Cotoneaster horizontalis Decne.	*Rosaceae*	Neo	Nat	Shrub	Asia
Cotoneaster salicifolius Franch.	*Rosaceae*	Arc	Nat	Shrub	Asia
Crassocephalum crepidioides (Benth.) S.Moore	*Asteraceae*	Neo	Nat	Herb	Africa
Cryptomeria japonica (Thunb. ex L.f.) D.Don	*Cupressaceae*	Neo	Cas	Tree	Asia
Cupressus arizonica Greene	*Cupressaceae*	Neo	Nat	Tree	America
Cupressus macrocarpa Hartw.	*Cupressaceae*	Neo	Nat	Tree	America
Cuscuta campestris Yunck.	*Cuscutaceae*	Neo	Nat	Herb	America
Cymbalaria muralis P.Gaertn., B.Mey. & Scherb.	*Plantaginaceae*	Arc	Nat	Herb	Mediterranean
*Cynoglossum wallichii var. glochidiatum (*Wall. ex Benth.) Kazmi	*Boraginaceae*	Arc	Nat	Herb	Asia
Cyperus congestus Vahl	*Cyperaceae*	Neo	Nat	Herb	Africa
Cyperus esculentus L.	*Cyperaceae*	Arc	Nat	Herb	Unknown
Cyperus rotundus L.	*Cyperaceae*	Arc	Nat	Herb	Eurasia
Dalbergia sissoo DC.	*Fabaceae*	Neo	Cas	Tree	Asia
Datura innoxia Mill.	*Solanaceae*	Neo	Nat	Herb	America
Datura metel L.	*Solanaceae*	Neo	Cas	Herb	Asia
Datura stramonium L.	*Solanaceae*	Neo	Nat	Herb	America
Deutzia gracilis Siebold & Zucc.	*Hydrangeaceae*	Arc	Nat	Shrub	Asia
Deutzia scabra Thunb.	*Hydrangeaceae*	Neo	Nat	Shrub	Asia
Dichondra repens J.R.Forst. & G.Forst.	*Convolvulaceae*	Neo	Cas	Herb	Asia
Dichrocephala integrifolia (L.f.) Kuntze	*Asteraceae*	Neo	Nat	Herb	Africa & Asia
Dieffenbachia seguine (Jacq.) Schott	*Araceae*	Neo	Nat	Herb	America
Digitaria sanguinalis (L.) Scop.	*Poaceae*	Neo	Nat	Herb	Europe & Africa
Diplachne fusca (L.) P.Beauv.	*Poaceae*	Neo	Nat	Herb	Unknown
Duchesnea indica (Jacks.) Focke	*Rosaceae*	Neo	Cas	Herb	Asia
Duranta erecta L.	*Verbenaceae*	Neo	Nat	Shrub/Tree	America
Dysphania ambrosioides (L.) Mosyakin & Clemants	*Amaranthaceae*	Neo	Nat	Herb	America
*Dysphania botrys (*L.) Mosyakin & Clemants	*Amaranthaceae*	Arc	Nat	Herb	Eurasia
Dysphania multifida (L.) Mosyakin & Clemants	*Amaranthaceae*	Neo	Nat	Herb	America
Echinochloa colonum (L.) Link	*Poaceae*	Neo	Nat	Herb	Unknown
*Echinochloa oryzoides (*Ard.) Fritsch	*Poaceae*	Arc	Nat	Herb	Asia
Echinopsis chamaecereus H.Friedrich & Glaetzle	*Cactaceae*	Neo	Nat	Succulent	America

Taxa	Family	Res	Stat	Simplified growth form	Native range
Egeria densa Planch.	*Hydrocharitaceae*	Neo	Nat	Aquatic	America
Eichhornia crassipes (Mart.) Solms	*Pontederiaceae*	Neo	Nat	Aquatic	America
Elatine ambigua Wight	*Elatinaceae*	Neo	Nat	Aquatic	Asia
Eleusine indica (L.) Gaertn.	*Poaceae*	Neo	Nat	Herb	Africa
Elodea canadensis Michx.	*Hydrocharitaceae*	Neo	Nat	Aquatic	America
Elsholtzia ciliata (Thunb.) Hyl.	*Lamiaceae*	Neo	Nat	Herb	Asia
Eragrostis curvula (Schrad.) Nees	*Poaceae*	Arc	Nat	Herb	Africa
Erigeron annuus (L.) Pers.	*Asteraceae*	Neo	Nat	Herb	America
Erigeron bonariensis L.	*Asteraceae*	Neo	Nat	Herb	America
Erigeron canadensis L.	*Asteraceae*	Neo	Nat	Herb	America
Erigeron sumatrensis Retz.	*Asteraceae*	Neo	Nat	Herb	America
Erythrina crista-galli L.	*Fabaceae*	Neo	Cas	Tree	America
Erythrina flabelliformis Kearney	*Fabaceae*	Neo	Cas	Tree	America
Eucalyptus camaldulensis Dehnh.	*Myrtaceae*	Neo	Cas	Tree	Australia
Eucalyptus grandis W.Hill	*Myrtaceae*	Neo	Cas	Tree	Australia
Euonymus fortunei (Turcz.) Hand.-Mazz.	*Celastraceae*	Arc	Nat	Shrub	Asia
Euonymus japonicus Thunb.	*Celastraceae*	Arc	Nat	Shrub/Tree	Asia
Eupatorium cannabinum L.	*Asteraceae*	Arc	Nat	Herb	Europe
Euphorbia chamaesyce L.	*Euphorbiaceae*	Neo	Nat	Herb	America
Euphorbia heterophylla L.	*Euphorbiaceae*	Neo	Cas	Herb	America
Euphorbia humifusa Willd.	*Euphorbiaceae*	Arc	Nat	Herb	Asia
Euphorbia lagascae Spreng.	*Euphorbiaceae*	Arc	Nat	Herb	Mediterranean
Euphorbia lathyris L.	*Euphorbiaceae*	Arc	Nat	Herb	Mediterranean
Euphorbia nutans Lag.	*Euphorbiaceae*	Neo	Nat	Herb	America
Euphorbia prostrata Aiton	*Euphorbiaceae*	Neo	Nat	Herb	America
Euphorbia serpens Kunth	*Euphorbiaceae*	Neo	Nat	Herb	America
Euphorbia supina Rafin.	*Euphorbiaceae*	Neo	Nat	Herb	America
Fallopia aubertii (L.Henry) Holub	*Polygonaceae*	Neo	Nat	Vine	Asia
*Fatsia japonica (*Thunb.) Decne. & Planch.	*Araliaceae*	Neo	Nat	Shrub/Tree	Asia
Ficus elastica Roxb. ex Hornem.	*Moraceae*	Neo	Cas	Tree	Asia
Ficus macrophylla Desf. ex Pers.	*Moraceae*	Neo	Cas	Tree	Australia
Ficus microcarpa L.f.	*Moraceae*	Neo	Cas	Tree	Asia
Forsythia × *intermedia* Zabel	*Oleaceae*	Neo	Cas	Shrub	Garden/Hybrid
Fragaria × *ananassa* (Duchesne ex Weston) Duchesne ex Rozier	*Rosaceae*	Neo	Cas	Herb	America
Gaillardia pulchella Foug.	*Asteraceae*	Neo	Cas	Herb	America
Galinsoga ciliata (Rafin) S.F. Blake	*Asteraceae*	Neo	Nat	Herb	America
Galinsoga parviflora Cav.	*Asteraceae*	Neo	Nat	Herb	America
Galinsoga quadriradiata Ruiz & Pav.	*Asteraceae*	Neo	Nat	Herb	America
Gasteria obliqua (Aiton) Duval	*Xanthorrhoeaceae*	Neo	Cas	Succulent	Africa
Gazania rigens (L.) Gaertn.	*Asteraceae*	Neo	Cas	Herb	Africa
Geranium pusillum L.	*Geraniaceae*	Neo	Nat	Herb	Eurasia
Gleditsia triacanthos L.	*Fabaceae*	Neo	Cas	Tree	America
Gomphocarpus fruticosus (L.) W.T.Aiton	*Apocynaceae*	Neo	Nat	Herb	Africa
Gypsophila elegans M.Bieb.	*Caryophyllaceae*	Arc	Nat	Herb	Eurasia

Taxa	**Family**	**Res**	**Stat**	**Simplified growth form**	**Native range**
Gypsophila pilosa Huds.	*Caryophyllaceae*	Arc	Nat	Herb	Asia
Heliotropium curassavicum L.	*Boraginaceae*	Neo	Nat	Herb	America
Hemerocallis fulva (L.) L.	*Hemerocallidaceae*	Neo	Nat	Herb	Asia
Hibiscus trionum L.	*Malvaceae*	Arc	Nat	Herb	Africa
Homalocladium platycladum (F.Muell.) L.H.Bailey	*Polygonaceae*	Neo	Cas	Shrub	Oceania
Hoya carnosa (L.f.) R.Br.	*Apocynaceae*	Neo	Cas	Vine	Asia
Hydrangea macrophylla (Thunb.) Ser.	*Hydrangeaceae*	Neo	Nat	Herb	Asia
Hydrocotyle ramiflora Maxim.	*Umbelliferae*	Neo	Nat	Aquatic	Asia
Imperata cylindrica (L.) Raeusch.	*Poaceae*	Neo	Nat	Herb	Asia
Ipomoea nil (L.) Roth	*Convolvulaceae*	Neo	Nat	Vine	America
Ipomoea purpurea (L.) Roth	*Convolvulaceae*	Neo	Nat	Vine	America
Ipomoea tricolor Cav.	*Convolvulaceae*	Neo	Nat	Vine	America
Ipomoea triloba L.	*Convolvulaceae*	Neo	Nat	Vine	America
Jacaranda mimosifolia D.Don	*Bignoniaceae*	Neo	Cas	Tree	America
Juncus tenuis Willd.	*Juncaceae*	Neo	Nat	Herb	America
Juniperus chinensis L.	*Cupressaceae*	Neo	Nat	Shrub/Tree	Asia
Juniperus horizontalis Moench	*Cupressaceae*	Neo	Nat	Shrub	America
Justicia brandegeeana Wassh. & L.B.Sm.	*Acanthaceae*	Neo	Cas	Shrub	America
Kalanchoe blossfeldiana Poelln.	*Crassulaceae*	Neo	Cas	Succulent	Africa (Madagascar)
Kerria japonica (L.) DC.	*Rosaceae*	Neo	Cas	Shrub	Asia
Kniphofia uvaria (L.) Oken	*Liliaceae*	Neo	Cas	Succulent	Africa
Koelreuteria paniculata Laxm.	*Sapindaceae*	Neo	Cas	Tree	Asia
Lagerstroemia indica L.	*Lythraceae*	Neo	Cas	Tree	Asia
Lantana camara L.	*Verbenaceae*	Neo	Cas	Shrub	America
Lepidium virginicum L.	*Brassicaceae*	Neo	Nat	Herb	America
Leucaena leucocephala (Lam.) de Wit	*Fabaceae*	Neo	Cas	Tree	America
Ligustrum ovalifolium Hassk.	*Oleaceae*	Neo	Cas	Shrub/Tree	Asia
Liquidambar styraciflua L.	*Altingiaceae*	Neo	Cas	Tree	America
Livistona mariae F.Muell.	*Arecaceae*	Neo	Cas	Palm	Australia
Lonicera japonica Thunb.	*Caprifoliaceae*	Neo	Cas	Vine	Asia
Lonicera ligustrina var. *yunnanensis* Franch.	*Caprifoliaceae*	Neo	Cas	Vine	Asia
Lonicera periclymenum L.	*Caprifoliaceae*	Neo	Nat	Vine	Europe & NW Africa
Ludwigia peploides (Kunth) P.H.Raven s.l.	*Onagraceae*	Neo	Cas	Aquatic	America
Lycianthes rantonnei (Carrière) Bitter	*Solanaceae*	Neo	Nat	Shrub	America
Lysimachia japonica Thunb.	*Primulaceae*	Neo	Nat	Herb	Asia
Maclura pomifera (Raf.) C.K.Schneid.	*Moraceae*	Neo	Nat	Tree	America
Magnolia grandiflora L.	*Magnoliaceae*	Neo	Cas	Tree	America
Malus floribunda Siebold ex Van Houtte	*Rosaceae*	Arc	Nat	Shrub/Tree	Asia
Matricaria discoidea DC.	*Asteraceae*	Neo	Nat	Herb	America
Matricaria matricarioides (Less.) Porter	*Asteraceae*	Neo	Nat	Herb	America
Melia azedarach L.	*Meliaceae*	Neo	Nat	Tree	Asia
Mesembryanthemum cordifolium L.f.	*Aizoaceae*	Neo	Nat	Succulent	Africa
Mesembryanthemum crystallinum L.	*Aizoaceae*	Neo	Nat	Succulent	Africa

Taxa	Family	Res	Stat	Simplified growth form	Native range
Mesembryanthemum nodiflorum L.	*Aizoaceae*	Arc	Nat	Succulent	Mediterranean & S Africa
Microstegium vimineum (Trin.) A.Camus	*Poaceae*	Neo	Nat	Herb	Asia
Mirabilis jalapa L.	*Nyctaginaceae*	Neo	Cas	Herb	America
Miscanthus sinensis Andersson	*Poaceae*	Neo	Cas	Bambusoid	Asia
Myriophyllum spicatum L.	*Haloragaceae*	Neo	Cas	Aquatic	Eurasia
Myriophyllum verticillatum L.	*Haloragaceae*	Neo	Cas	Aquatic	Circumboreal
Nandina domestica Thunb.	*Berberidaceae*	Neo	Cas	Bambusoid	Asia
Nephrolepis exaltata (L.) Schott	*Nephrolepidaceae*	Neo	Cas	Fern	America
Nicotiana glauca Graham	*Solanaceae*	Neo	Nat	Shrub/Tree	America
Oenothera biennis L.	*Onagraceae*	Neo	Nat	Herb	America
Oenothera glazioviana Micheli	*Onagraceae*	Neo	Nat	Herb	Garden/Hybrid
Oenothera parodiana Munz	*Onagraceae*	Neo	Nat	Herb	America
Oldenlandia capensis L.f. var. *capensis*	*Rubiaceae*	Neo	Nat	Herb	Africa
Oldenlandia capensis var. *pleiosepala* Bremek.	*Rubiaceae*	Neo	Cas	Herb	Africa
Opuntia ficus-indica (L.) Mill.	*Cactaceae*	Neo	Nat	Succulent	America
Opuntia microdasys (Lehm.) Pfeiff.	*Cactaceae*	Neo	Nat	Succulent	America
Oryza sativa L.	*Poaceae*	Arc	Cas	Herb	Asia
Oxalis articulata Savigny	*Oxalidaceae*	Neo	Nat	Herb	America
Oxalis corniculata L. s.l.	*Oxalidaceae*	Arc	Nat	Herb	America
Oxalis debilis var. *corymbosa* (DC.) Lourteig	*Oxalidaceae*	Neo	Nat	Herb	America
Oxalis floribunda Lehm.	*Oxalidaceae*	Neo	Nat	Herb	America
Oxalis pes-caprae L.	*Oxalidaceae*	Neo	Nat	Herb	Africa
Oxalis pes-caprae f. *pleniflora* (Lowe) Sunding	*Oxalidaceae*	Neo	Nat	Herb	Africa
Oxalis stricta L.	*Oxalidaceae*	Neo	Nat	Herb	America
Panicum capillare L.	*Poaceae*	Neo	Nat	Herb	America
Panicum miliaceum L.	*Poaceae*	Arc	Nat	Herb	Asia
Parkinsonia aculeata L.	*Fabaceae*	Neo	Cas	Tree	America
Parthenocissus quinquefolia (L.) Planch.	*Vitaceae*	Neo	Cas	Vine	America
Paspalum dilatatum Poir.	*Poaceae*	Neo	Nat	Herb	America
Paspalum distichum L.	*Poaceae*	Neo	Nat	Herb	America
Paspalum thunbergii Kunth ex Steud	*Poaceae*	Arc	Cas	Herb	Asia
Passiflora caerulea L.	*Passifloraceae*	Neo	Cas	Vine	America
Paulownia elongata S. Y. Hu.	*Paulowniaceae*	Neo	Cas	Tree	Asia
Paulownia fortunei (Seem.) Hemsl.	*Paulowniaceae*	Neo	Cas	Tree	Asia
Paulownia fortunei x Paulownia tomentosa	*Paulowniaceae*	Neo	Cas	Tree	Garden/Hybrid
Paulownia tomentosa Steud.	*Paulowniaceae*	Neo	Cas	Tree	Asia
Pelargonium zonale (L.) L'Hér. ex Aiton	*Geraniaceae*	Neo	Nat	Shrub	Africa
Perilla frutescens (L.) Britton	*Lamiaceae*	Neo	Cas	Herb	Asia
Phacelia tanacetifolia Benth.	*Hydrophyllaceae*	Neo	Cas	Herb	America
Phaseolus vulgaris L.	*Fabaceae*	Neo	Cas	Vine	America
Phyla canescens (Kunth) Greene	*Verbenaceae*	Neo	Nat	Herb	America
Phyla nodiflora (L.) Greene	*Verbenaceae*	Neo	Nat	Herb	America
Phyllostachys bambusoides Siebold & Zucc.	*Poaceae*	Neo	Nat	Bambusoid	Asia
Physalis alkekengi L. s.l.	*Solanaceae*	Neo	Nat	Herb	Eurasia

Taxa	**Family**	**Res**	**Stat**	**Simplified growth form**	**Native range**
Physalis angulata L.	*Solanaceae*	Neo	Nat	Herb	America
Physalis philadelphica var. *immaculata* Waterf.	*Solanaceae*	Neo	Nat	Herb	America
Physalis pubescens L.	*Solanaceae*	Neo	Nat	Herb	America
Phytolacca americana L.	*Phytolaccaceae*	Neo	Nat	Herb	America
Picea glauca (Moench) Voss	*Pinaceae*	Neo	Nat	Tree	America
Pinus pinaster Aiton	*Pinaceae*	Arc	Nat	Tree	Mediterranean
Pinus ponderosa Douglas ex C.Lawson	*Pinaceae*	Neo	Nat	Tree	America
Pinus radiata D.Don	*Pinaceae*	Neo	Cas	Tree	America
Pittosporum tobira (Thunb.) W.T.Aiton	*Pittosporaceae*	Neo	Cas	Shrub	Asia
Platycladus orientalis (L.) Franco	*Cupressaceae*	Neo	Nat	Tree	Asia
Plumbago auriculata Lam.	*Plumbaginaceae*	Neo	Cas	Shrub	Africa
Polygala myrtifolia L.	*Polygalaceae*	Neo	Cas	Shrub	Africa
Polygonum perfoliatum L.	*Polygonaceae*	Neo	Nat	Vine	Asia
Polygonum thunbergii Siebold & Zucc.	*Polygonaceae*	Arc	Nat	Herb	Asia
Populus × canadensis Moench	*Salicaceae*	Neo	Nat	Tree	Garden/Hybrid
Populus deltoides Bartr. ex Marsh.	*Salicaceae*	Neo	Nat	Tree	America
Portulaca grandiflora Hook.	*Portulacaceae*	Neo	Cas	Herb	America
Portulaca oleracea L. s.l.	*Portulacaceae*	Arc	Nat	Herb	Mediterranean
Pseudosasa japonica (Steud.) Makino	*Poaceae*	Neo	Cas	Bambusoid	Asia
Pseudotsuga menziesii (Mirb.) Franco var. *menziesii*	*Pinaceae*	Neo	Cas	Tree	America
Pseudotsuga menziesii (Mirb.) Franco var. *glauca* (Beissn.) Franco	*Pinaceae*	Neo	Cas	Tree	America
Quercus rubra L.	*Fagaceae*	Neo	Cas	Tree	America
Rhapis excelsa (Thunb.) Henry	*Arecaceae*	Neo	Nat	Palm	Asia
Ricinus communis L.	*Euphorbiaceae*	Arc	Nat	Shrub	Africa
Robinia hispida L.	*Fabaceae*	Neo	Cas	Tree	America
Robinia pseudoacacia L.	*Fabaceae*	Neo	Nat	Tree	America
Rudbeckia hirta L.	*Asteraceae*	Neo	Cas	Herb	America
Russelia equisetiformis Schltdl. & Cham.	*Plantaginaceae*	Neo	Cas	Shrub	America
Salix babylonica L.	*Salicaceae*	Neo	Nat	Tree	Asia
Santolina chamaecyparissus L.	*Asteraceae*	Arc	Nat	Herb	Mediterranean
Saponaria officinalis L.	*Caryophyllaceae*	Arc	Nat	Herb	Eurasia
Schefflera arboricola (Hayata) Merr.	*Araliaceae*	Neo	Cas	Shrub	Asia
Schinus molle L.	*Anacardiaceae*	Neo	Cas	Tree	America
Schinus terebinthifolius Raddi	*Anacardiaceae*	Neo	Cas	Tree	America
Scopolia carniolica Jacq.	*Solanaceae*	Arc	Nat	Herb	Europe
Sequoia sempervirens (D.Don) Endl.	*Cupressaceae*	Neo	Cas	Tree	America
Sequoiadendron giganteum (Lindl.) J.Buchholz	*Cupressaceae*	Neo	Cas	Tree	America
Setaria faberi R.A.W.Herrm.	*Poaceae*	Neo	Nat	Herb	Asia
Setaria italica (L.) P.Beauv.	*Poaceae*	N/A	Nat	Herb	Unknown
Setaria viridis (L.) P.Beauv.	*Poaceae*	Neo	Nat	Herb	Eurasia
Sicyos angulatus L.	*Cucurbitaceae*	Neo	Nat	Vine	America
Sida spinosa L.	*Malvaceae*	Neo	Nat	Herb	America
Sigesbeckia pubescens (Makino) Makino	*Asteraceae*	Neo	Cas	Herb	Asia
Solanum americanum Mill.	*Solanaceae*	N/A	Nat	Herb	Unknown

Taxa	Family	Res	Stat	Simplified growth form	Native range
Solanum angustifolium Mill.	*Solanaceae*	Neo	Cas	Herb	America
Solanum elaeagnifolium Cav.	*Solanaceae*	Neo	Nat	Herb	America
Solanum jasminoides J.Paxton	*Solanaceae*	Neo	Cas	Vine	America
Solanum luteum Mill. s.l.	*Solanaceae*	N/A	Nat	Herb	Mediterranean & E Asia
Solanum lycopersicum L.	*Solanaceae*	Neo	Cas	Herb	America
Solanum pseudocapsicum L.	*Solanaceae*	Neo	Cas	Herb	America
Solanum pseudocapsicum var. *diflorum* (Vell.) Bitter	*Solanaceae*	Neo	Cas	Herb	America
Solanum sisymbriifolium Lam.	*Solanaceae*	Neo	Nat	Herb	America
Solanum sodomaeum L.	*Solanaceae*	Neo	Nat	Shrub	Africa
Solanum tuberosum L.	*Solanaceae*	Neo	Cas	Herb	America
Solidago canadensis L.	*Asteraceae*	Neo	Nat	Herb	America
Sorghum × *drummondii* (Nees ex Steud.) Millsp. & Chase	*Poaceae*	Neo	Cas	Bambusoid	Garden/Hybrid
Sorghum bicolor (L.) Moench	*Poaceae*	Arc	Cas	Bambusoid	Africa
Spiraea × *vanhouttei* (Briot) Zabel	*Rosaceae*	Neo	Cas	Shrub	Garden/Hybrid
Sporobolus fertilis (Steud.) Clayton	*Poaceae*	Neo	Nat	Herb	Asia
Sporobolus indicus (L.) R.Br.	*Poaceae*	Neo	Nat	Herb	America
Strelitzia reginae Banks	*Strelitziaceae*	Neo	Cas	Herb	Africa
Styphnolobium japonicum (L.) Schott	*Fabaceae*	Neo	Cas	Tree	Asia
Symphyotrichum laeve (L.) Á.Löve & D.Löve	*Asteraceae*	Neo	Nat	Herb	America
Symphyotrichum squamatum (Spreng.) G.L.Nesom	*Asteraceae*	Neo	Nat	Herb	America
Syringa vulgaris L.	*Oleaceae*	Neo	Nat	Shrub	Europe
Tagetes erecta L.	*Asteraceae*	Neo	Nat	Herb	America
Tagetes minuta L.	*Asteraceae*	Neo	Nat	Herb	America
Tecoma capensis (Thunb.) Lindl.	*Bignoniaceae*	Neo	Cas	Vine	Africa
Thuja plicata Donn ex D.Don	*Cupressaceae*	Neo	Nat	Tree	America
Tradescantia fluminensis Vell.	*Commelinaceae*	Neo	Nat	Herb	America
Tradescantia pallida (Rose) D.R.Hunt	*Commelinaceae*	Neo	Cas	Herb	America
Tropaeolum majus L.	*Tropaeolaceae*	Neo	Nat	Vine	America
Ulex europaeus L.	*Fabaceae*	Neo	Nat	Shrub	Europe
Veronica persica Poir.	*Plantaginaceae*	Neo	Nat	Herb	Asia
Vinca minor L.	*Apocynaceae*	Arc	Nat	Herb	Europe
Vitis riparia Michx s.l.	*Vitaceae*	Neo	Cas	Vine	America
Washingtonia robusta H.Wendl.	*Arecaceae*	Neo	Cas	Palm	America
Weigela florida (Bunge) A.DC.	*Caprifoliaceae*	Neo	Nat	Shrub	Asia
Wisteria sinensis (Sims) Sweet	*Fabaceae*	Neo	Nat	Vine	Asia
Withania somnifera (L.) Dunal	*Solanaceae*	Arc	Nat	Shrub	Asia
Xanthium spinosum L.	*Asteraceae*	Neo	Nat	Herb	America
Xanthium strumarium L. s.l.	*Asteraceae*	Arc	Nat	Herb	America
Yucca gloriosa L.	*Asparagaceae*	Neo	Cas	Succulent	America
Zantedeschia aethiopica (L.) Spreng.	*Araceae*	Neo	Cas	Herb	Africa
Zizyphus mauritiana Lamk.	*Rhamnaceae*	Arc	Nat	Shrub/Tree	Asia

Supplementary material 1

Alien flora of Turkey: checklist, taxonomic composition and ecological attributes

Authors: Ahmet Uludağ, Necmi Aksoy, Ayşe Yazlık, Zübeyde Filiz Arslan, Efecan Yazmış, İlhan Üremiş, Tiziana Antonella Cossu, Quentin Groom, Jan Pergl, Petr Pyšek, Giuseppe Brundu

Data type: List of alien plants

Explanation note: List of alien taxa in the flora of Turkey. Taxa are ordered alphabetically. Each taxon is listed together with its family, residence time, invasion status, life-form according to Raunkiaer, growth for according to the Thesaurus of Plant Characteristics for Ecology and Evolution, simplified growth-form, life history, reasons for intentional and accidental introduction. The last five columns on the right list habitats where the species is found in Turkey. This list includes also 47 frequently planted taxa.

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RESEARCH ARTICLE

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The prioritisation of a short list of alien plants for risk analysis within the framework of the Regulation (EU) No. 1143/2014

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Abstract

Thirty-seven alien plant species, pre-identified by horizon scanning exercises were prioritised for pest risk analysis (PRA) using a modified version of the EPPO Prioritisation Process designed to be compliant with the EU Regulation 1143/2014. In Stage 1, species were categorised into one of four lists – a Residual List, EU List of Minor Concern, EU Observation List and the EU List of Invasive Alien Plants. Only those species included in the latter proceeded to the risk management stage where their priority for PRA was assessed. Due to medium or high spread potential coupled with high impacts twenty-two species were included in the EU List of Invasive Alien Plants and proceeded to Stage 2. Four species (*Ambrosia trifida*, *Egeria densa*, *Fallopia baldschuanica* and *Oxalis pes-caprae*) were assigned to the EU Observation List due to moderate or low impacts. *Albizia lebbeck*, *Clematis terniflora*, *Euonymus japonicus*, *Lonicera morrowii*, *Prunus campanulata* and *Rubus rosifolius* were assigned to the residual list due to a current lack of information on impacts. Similarly, *Cornus sericea* and *Hydrilla verticillata* were assigned to the Residual List due to unclear taxonomy and uncertainty in native status, respectively. *Chromolaena odorata*, *Cryptostegia grandiflora* and *Sphagneticola trilobata* were assigned to the Residual List as it is unlikely they will establish in the Union under current climatic conditions. In the risk management stage, *Euonymus fortunei*, *Ligustrum sinense* and *Lonicera maackii* were considered a low priority for PRA as they do not exhibit invasive tendencies despite being widely cultivated in the EU over several decades. Nineteen species were identified as having a high priority for a PRA (*Acacia dealbata*, *Ambrosia confertiflora*, *Andropogon virginicus*, *Cardiospermum grandiflorum*, *Celastrus orbiculatus*, *Cinnamomum camphora*, *Cortaderia jubata*, *Ehrharta calycina*, *Gymnocoronis spilanthoides*, *Hakea sericea*, *Humulus scandens*, *Hygrophila polysperma*, *Lespedeza cuneata*, *Lygodium japonicum*, *Pennisetum setaceum*, *Prosopis juliflora*, *Sapium sebiferum*, *Pistia stratiotes* and *Salvinia molesta*).

Keywords

Biodiversity, ecosystem services, Europe, impact, non-native, risk management

Introduction

Trade liberalisation and rapid globalisation have led to the increased spread of invasive alien species (IAS) around the world (van Kleunnen et al. 2015). IAS (plants, animals, fungi or micro-organisms) are recognised as one of the greatest threats to biological diversity by inflicting irreversible damage to the ecosystems they invade (Wilcove et al. 1998). In Europe, there are an estimated 12,000 alien species with 10-15 % considered invasive and it is these species that cost the EU around €12-billion per year (European Commission 2014, Kettunen et al. 2008).

Established invasive alien plant species are one of the largest groups of IAS both in terms of species numbers and the area they occupy (Sheppard et al. 2006). There are an estimated 3,749 naturalised alien plant species in Europe of which 1,780 are alien to Europe, with the remaining being native to parts of Europe (Pyšek et al. 2009). When alien plants invade regions, they can outcompete native plant species through direct (Daehler 2003) or indirect competition (Murrell et al. 2011). Impacts, because of habitat modification and displacement of native plant species can cascade to higher trophic levels impacting at an ecosystem scale (Tanner et al. 2013, Daniel et al. 2003, Levine et al. 2003). Although impacts on ecosystem services are less studied, examples show negative effects on provisioning (Kasulo 2000, Eagle et al. 2007), regulating (Chittka and Schürkens 2001, Prater et al. 2006) and cultural services (Chilton et al. 2002, McFarland et al. 2004).

To mitigate the threat of IAS to the European Union (EU), the European Commission adopted the EU Regulation (No. 1143/2014) 'on the prevention and management of the introduction and spread of invasive alien species' which came into force on the 1st January 2015 (EU 2014, Genovesi et al. 2015, Tollington et al. 2015). The EU Regulation, hereafter referred to as the IAS Regulation, aims to primarily address the negative impact of IAS on biodiversity and ecosystem services, while impacts on human health and the economy are considered as aggravating factors. The

IAS Regulation is centred around three main themes (1) prevention, (2) early warning and rapid response, and (3) management. The IAS Regulation will restrict the use, trade and transport of certain IAS and will be underpinned by a list of IAS of Union concern. At present the Union List contains 37 IAS, of which 14 species are invasive alien plants (European Commission 2016).

The IAS of Union concern will be subject to stringent enforcements including a ban on sale and preventative actions such as a ban on import (see Genovesi et al. 2015). Member States will be obliged to prevent the spread and conduct eradication and management measures for species on the list and already present in Member States (EU 2014). In theory, such measures would go a long way to mitigating entry and impacts of invasive, or potentially invasive alien plants in the EU, especially when considering two thirds of established alien plant species have been introduced intentionally for horticulture or agricultural purposes (Keller et al. 2011).

The IAS Regulation places an emphasis on prevention as opposed to cure, and as such the focus should be on species with a limited regional distribution within the Union, and species that are currently absent but pose a potential threat in the future. Many European countries and regional organisations have produced species lists and conducted horizon scanning studies which have identified priority species (Gallardo et al. 2015, Roy et al. 2015). However, for a species to be included in the list of Union concern a risk assessment is required to technically and objectively evaluate scientific and economic evidence to determine the level of risk associated with a species. It should be noted that the European and Mediterranean Plant Protection Organization (EPPO) always combines risk assessment with risk management, resulting in a risk analysis and hereafter referred to as a pest risk analysis (PRA).

A PRA can be a time-consuming process requiring significant finances and high levels of species specific expertise. When presented with a large pool of invasive, or potentially invasive alien plants, prioritizing species for PRA is an essential prerequisite to focus limited resources. High priority species would be those that have the highest negative impact and can be prevented from entering, or cost effectively managed in the European Union (Kumschick et al. 2012, Branquart et al. 2016).

Several schemes have been developed for different countries or regions to prioritise alien plants (Austria-Germany: Essl et al. 2011, Belgium: D'hondt et al. 2014, central Europe: Weber and Gut 2004). The scheme by Brunel et al. (2010) was designed to assess alien plants under the Plant Health Regulation. However, in the context of the IAS Regulation, more emphasis is required on impacts on biodiversity and ecosystem services. Due to this shift in the regulatory process of invasive alien plants, a new prioritisation scheme was designed to ensure that species prioritisation was compliant with the IAS Regulation (Branquart et al. 2016). What the new prioritisation process allows is to (1) prioritise species based on their impacts and spread, (2) to exclude species unsuited for PRA due to a lack of scientific information and (3), include the effectiveness of potential risk management measures for a given species in the prioritisation process. Thus, the prioritisation process deals with both risk assessment and risk management (i.e. risk analysis).

The objective of this study was to produce a list of alien plant species that comply with the definitions and criteria of Article 4 of the IAS Regulation, i.e. alien species that would be capable of causing major detrimental impacts on biodiversity and associated ecosystem services after establishment and spread within the EU territory, and to determine which of these have the highest priority for PRA at the European level.

Method

In March 2016, a three-day workshop was held at EPPO in Paris (FR), with the purpose of prioritising a list of invasive alien plants for PRA as part of a LIFE funded project 'Mitigating the threat of invasive alien plants in the EU through pest risk analysis to support the EU Regulation 1143/2014' (LIFE15 PRE FR 001) (see, www.IAP-risk.eu). Eight experts from the EPPO Panel on Invasive Alien Plants, the NERC Centre for Ecology and Hydrology and the EPPO Secretariat attended the workshop.

Species selected for prioritisation

We appreciate that there are numerous alien plants which could be proposed as candidates for prioritisation, however, due to limited time and financial resources we focused on species that had already been preselected by horizon scanning from two sources. Species were taken from the EPPO List of Invasive Alien Plants (see www.eppo.int) and a recent horizon scanning exercise by Roy et al. (2015).

The EPPO List of Invasive Alien Plants included a total of 15 plant species identified as having a high priority for a PRA whereas Roy et al. (2015) identified a total of 24 plant species which present a high or very high risk to the EU within the next ten years. Of the 24 species identified in Roy et al. (2015), two species (*Alternanthera philoxeroides* (Mart.) Griseb. and *Microstegium vimineum* (Trin.)) had recently been risk analysed (see www.eppo.int) and were excluded from further assessment. Therefore, 22 species from Roy et al. (2015) and 15 species from the EPPO List of Invasive Alien Plants were combined to produce a list of 37 species for prioritisation. Further prioritisation of these 37 species was required based on the requirements of the IAS Regulation. In the case of the species from the EPPO list, these species were selected using the original EPPO prioritisation Scheme (Brunel et al., 2010), where the focus for selection was based on the criteria of the Plant Health Regulation. The species from Roy et al. (2015) included species where scientific data (e.g. impacts, establishment etc.) was lacking, and in addition, European Union outermost regions (e.g. Azores, Canary Islands and Madeira) were included as areas at risk though the IAS Regulation excludes these regions. Lastly, when Roy et al. (2015) prioritised their species risk management criteria were not considered. We suggest that risk management is a vital consideration when prioritizing species for the IAS Regulation to select species where preventative actions are feasible (see Article 4.3 (e) and Article 4.6).

EPPO prioritisation process compliant with the IAS Regulation

The prioritisation scheme used for this study was an amended version of the EPPO prioritisation process for Invasive Alien Plants (Brunel et al. 2010, EPPO standard PM5/6), specifically adapted within the remit of the LIFE project to be fully compliant with the IAS Regulation. A full description of the process is given in Branquart et al. (2016) and depicted in figure (1). The prioritisation process was designed to meet the requirements of Article 4 (IAS Regulation) where the highest priority for performing a PRA at the European level is given to species that satisfy the following criteria: (i) they are alien to the territory of the EU excluding the outermost regions, (ii) they are capable of establishing a viable population and spreading rapidly in the environment in the EU (excluding the outermost regions), (iii) they are capable of causing major detrimental impacts to biodiversity and the associated ecosystem services, (iv) actions can be taken to effectively prevent, minimise or mitigate their adverse impact, which involves that they are moved from country to country primarily by human activities and they still have a significant area suitable for further spread within the EU (EU 2014).

The first stage of the process, the preliminary risk assessment stage, categorises each species into one of four lists (Residual List, EU List of Minor Concern, EU Observation List and the EU List of Invasive Alien Plants) by addressing pre-determined criteria (questions). To proceed to any of the three EU lists, each species needs to meet the requirements of questions A1, A2, A3, A5 and A6, i.e. a positive (yes) answer is required. If a negative (no) answer is recorded, the species is included in the Residual List of species that do not qualify. Reasons a species (including subspecies, varieties, hybrids and cultigens, hereafter collectively called species) may be included in the Residual list include uncertainty in taxonomy and nomenclatural (question A1. Fig. 1), or a lack of current scientific information (question A3. Fig.1). Only those species included in the EU List of Invasive Alien Plants (those species which have the highest potential spread capacity and high negative impacts on biodiversity or ecosystem services) proceed to the risk management stage.

Within the second stage, the preliminary risk management stage, priority for a PRA at the EU level is evaluated based on the feasibility and cost-effectiveness of mitigating impacts with management measures and/or preventative actions. The output of stage two is to define the species into one of two categories:

(1) the plant species is included in a List of Priority Invasive Alien Plants for performing an EU level PRA,
(2) the plant species is included in a List of Invasive Alien Plants that are not considered as a priority to conduct a EU level PRA.

Gathering of species information

Scientific information was collected for each species prior to the workshop. Each expert collected detailed scientific information on each species from a number of predeter-

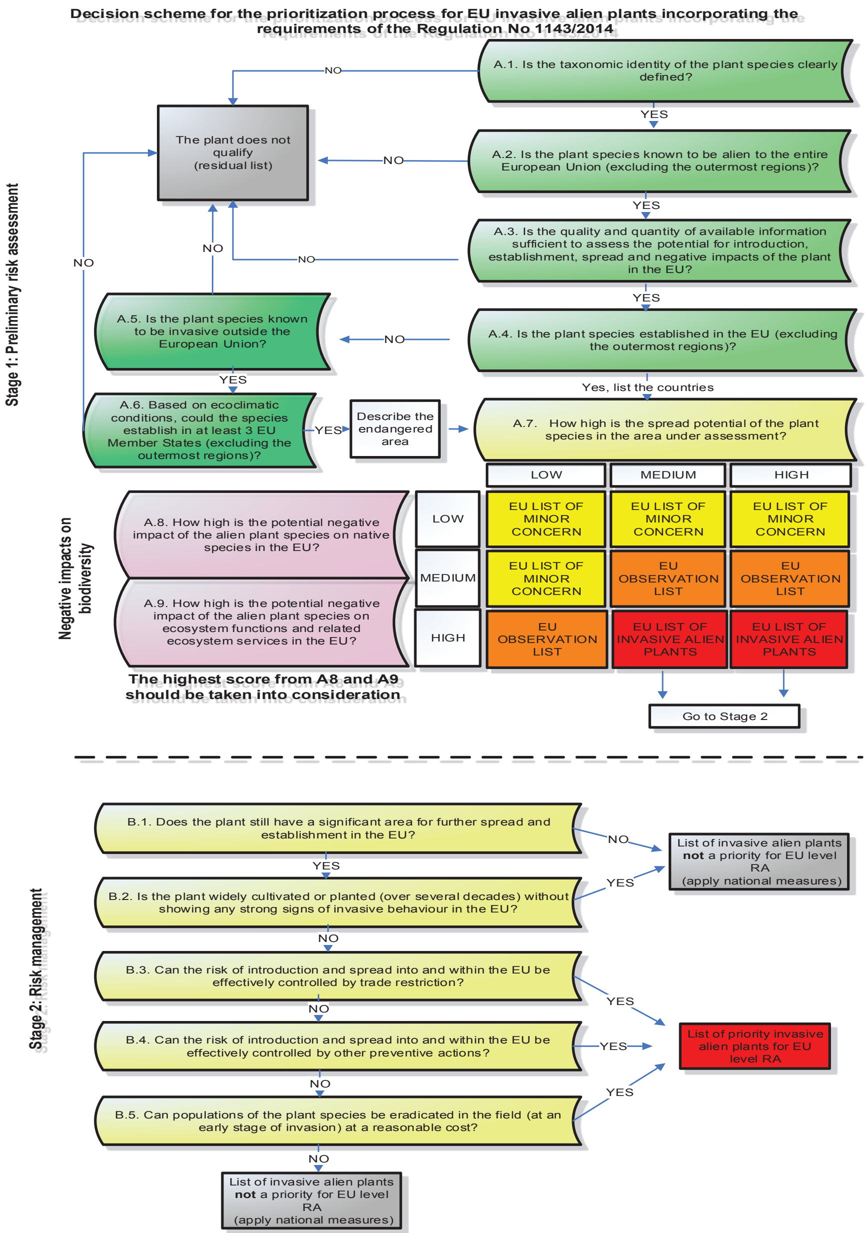

Figure 1. Decision scheme for the EU prioritisation process for alien plants (Taken from Branquart et al. 2016).

species occurrence were developed. A key criterion in evaluating the risk of a species to the EU is to assess if the species can establish under current climatic conditions. This is especially important for species which are currently absent from the region but have been highlighted as a risk through horizon scanning exercises.

Table 1. Key information sources. Information resources utilised when collecting information on the species.

Scientific area	Relating to question in EU P. process	Key resources
Stage 1		
Taxonomic identity	A1	The Plant List (http://www.theplantlist.org/)
Geographical origin	A2	ARS Grin Taxonomy (http://www.ars-grin.gov/)
Global occurrence	A4	GBIF (http://www.gbif.org), EPPO Global Database (https://gd.eppo.int/), CABI ISC (http://www.cabi.org/isc/), Q-Bank (http://www.q-bank.eu/)
Global invasive behavior	A5	Scientific literature, reports, expert opinion
Spread potential & areas threatened	A6, A7	Scientific literature, reports, expert opinion
Impacts	A8, A9	Scientific literature, reports, expert opinion
Stage 2		
Current occurrence within the EU	B1	GBIF (http://www.gbif.org), EPPO Global Database (https://gd.eppo.int/), CABI ISC (http://www.cabi.org/isc/), Q-Bank (http://www.q-bank.eu/)
Invasive behavior in the EU	B2	Scientific literature, reports, expert opinion
Trade status	B3	Numerous internet suppliers (e.g. https://www.rhs.org.uk/; http://www.ebay.com/; https://www.amazon.com/)
Phytosanitary measures	B4, B5	Scientific literature, reports, expert opinion

mined resources, including online databases scientific publications (internet searches and Web of Science), grey literature and relevant books and personal communications (see Table 1). For each species, where possible, the primary data sources were reviewed.

Quality and quantity of information for each species was evaluated under the main headings set out in Table 1. Quantitative data from scientific publications (scientific papers and reports) was considered superior to unreferenced information gathered from online databases. However, during the prioritisation assessment, all information was included and where unreferenced information was considered important, a concerted effort was taken to substantiate any reports. Each species was prioritised using compiled information where each question was answered in chronological order (see Figure 1). A consensus was reached between the experts based on available information and expert opinion.

Uncertainty scores were assigned to questions A7 (spread) and A8–A9 (impacts) following the criteria set out in Branquart et al. (2016). Uncertainty scores increase where the species is absent from the EU or information on a species was conflicting.

Modelling the potential occurrence of species

To support question A6, 'based on ecoclimatic conditions, could the species establish in at least 3 EU Member States (excluding the outermost regions)', maps of potential

However, modelling the potential distributions of alien species presents challenges, including the non-equilibrium nature of the distribution, presence of casual records representing failed introductions and spatial biases in recording effort (Václavik and Meentemeyer 2012). Substantial effort is usually required to develop accurate models that account for these effects, prohibiting the use of such models for rapid multi-species PRA prioritisation exercises. Therefore, we adopted a simple but precautionary approach based on delimiting a 'climate envelope' of each species that can be projected onto a map of Europe.

To delimit climate envelopes, we used the 19 standard bioclimatic variables gridded at 10 arcminute resolution (0.167 × 0.167 decimal degrees) from WorldClim (Hijmans et al. 2005). 'Climate space' was summarised by taking the first two axes of a principal components analysis (PCA) on centred and scaled bioclimatic variables, with log-transformed precipitation variables. These axes captured 77.5 % of the variation in climate. For each species, georeferenced occurrence data was obtained from the Global Biodiversity Information facility (GBIF: www.gbif.org). Data points were filtered according to expert opinion (Figure 2A). The species occurrences were then plotted in climate space, by extracting the PCA axis scores for occurrence locations (Figure 2B). To delimit a climate envelope for each species, bivariate density kernels were fitted to the occurrences in climate space using the kernelUD function of R package adehabitat with automatic selection of the smoothing parameter (Calenge 2006). From these models, 95 % kernel density polygons were extracted for each species. These bound the region of climate space containing 95 % of the smoothed occurrence density of each species. Finally, the climate envelopes were projected onto the EU by identifying the grid cells whose PCA axis scores fell inside the species' climate envelope (Figure 2C). The resulting maps were critically appraised by the working group panel, using their expert knowledge to consider the accuracy of the estimates and the potential for non-climatic factors such as habitat availability to limit establishment.

We emphasise that this method does not provide a definitive estimate of the potential for further species establishment, but rather a way of rapidly assessing if a species is worthy of further consideration in full PRA. We also note that the 95 % density kernels may be overly generous and exceed the climatic tolerances of the species. However, while a lower percentage threshold could have been used to constrict the envelopes, a precautionary approach is desirable for our purpose, given that invasive species may not have fully filled their climate niche space and because many species can invade outside of their native climatic niche (Bocsi et al. 2016).

Results

The 37 alien plant species prioritised in this study include representatives from 23 families where Asteraceae (5 species) and Poaceae (4 species) are most represented (Table 1). In total, the list contained 6 aquatics and 31 terrestrial species. Terrestrial

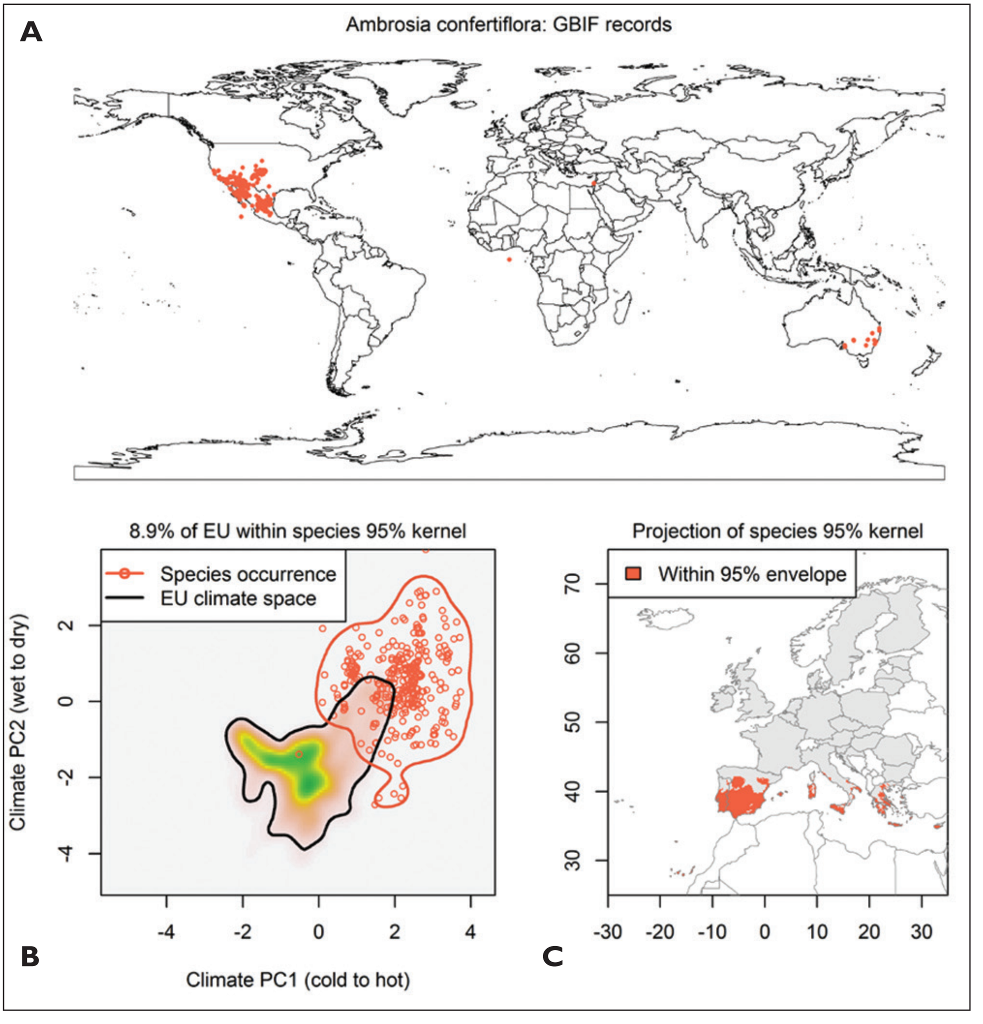

Figure 2. An example of the distribution maps and potential occurrence in Europe – *Ambrosia confertiflora*. **A** Global occurrence locations were obtained from the Global Biodiversity Information Facility **B** The global climate was summarised as two principal components analysis (PCA) axes on the 19 WorldClim layers (Hijmans et al. 2005). Species occurrences were plotted in this climate space and a bivariate normal kernel density model (Calenge 2006) was used to estimate 'climate envelopes' at different percentiles **C** These envelopes were then projected onto geographic space in the EU. Shading indicates these percentiles, with smaller numbers indicating higher density of occurrences.

species included 4 perennial grasses, 10 vines, 6 tree species, 7 woody shrubs and 4 perennial herbs. Almost half of the species (43 %) were native to Asia, followed by South America (18 %), North America (13 %), Africa (8 %), Australia (5 %) and pan-global species (8 %).

Stage 1 (risk assessment)

The first stage of the prioritisation process categorised 22 plant species in the EU List of Invasive Alien Plants, 4 plant species in the EU Observational List and 11 species in the Residual List (Table 2). None of the species were assigned to the List of Minor Concern. All species assigned to the EU List of Invasive Alien Plants fulfilled the criteria set out in questions A1 to A3; indicating a clear taxonomy, alien to the EU and the quality of information was sufficient to assess traits and impacts.

Cornus sericea L. did not fulfil the criteria of the first question in the prioritisation process 'Is the taxonomic identity of the plant species clearly established' as naturalised plants belong to a complex of hybrids of *C. sericea* and *C. alba* (Q-Bank 2016) and thus was included in the Residual List. Similarly, *Hydrilla verticillata* (L.f.) Royle was included in the Residual List as there is evidence the species is native in the EU (Ireland, Poland and the Baltic states; Cook and Lüönd 1982). *Albizia lebbeck* (L.) Benth., *Clematis terniflora* DC., *Euonymus japonicus* Thunb., *Lonicera morrowii* A. Gray, *Prunus campanulata* Maxim. and *Rubus rosifolius* SM. were assigned to Residual List as the quality of information for each was insufficient, potentially impeding a consise PRA.

Of the 29-species assessed under question A4 (is the plant species established in the EU excluding the outermost regions?), 68 % are recorded as established (Table 2). However, this includes 12 species where a clear established population could be debated, and for these species questions A5 and A6 were answered for completeness. All species were invasive in at least one geographical region in the world (excluding the EU), though 50 % of the species are recorded as invasive in two geographical regions, 13 % in three geographical regions and one species *Pistia stratiotes* L. is recorded as invasive in four regions.

Three species, *Chromolaena odorata* (L.) King & H.E.Robins, *Cryptostegia grandiflora* R.Br. and *Sphagneticola trilobata* (L.) Pruski were assigned to the Residual List due to uncertainty in potential for establishment (A.6). Species occurrence maps overlaid in EU climate space indicated establishment at 0.1 %, 0.3 % and 0.2 %, respectively.

The majority of species evaluated in question A7 (88 %) were assigned a high score for spread potential, indicating the species is highly fecund and propagules can spread over distances of 500 to 1,000 m from the parent plant (Table 3). Except for the aquatic species, all species are vigorous seed producers with evidence that propagules are carried by wind, water or wildlife (see Table 3).

For impact (A8: impacts on native plant species, A9: impacts on ecosystem functions and related ecosystem services), the highest of the two scores from A8 and A9 was used in the assessment. A high impact score, coupled with a medium or high spread potential, categorised the species in the EU List of Invasive Alien Plants whereas a medium impact score, coupled with a medium or high spread potential, listed the species in the EU Observation List. It is interesting to note that 84 % of species assessed in question A8 scored high compared to only 19 % scoring high for impacts on ecosystem functions and related ecosystem services. The low percentages for the latter may reflect the current lack of data on such impacts compared to direct impacts on native

Table 2. Results from the prioritisation exercise (Stage 1: risk assessment). The first stage of the prioritisation process categorised twenty-two plant species in the European List of Invasive Alien Plants (List IAP), 4 plant species in the European Observational List (Obs List) and 11 species were rejected from the process (Residual List) as they did not fulfil the criteria of specific questions. Area abbreviations follow: Aus: Australia, N. Am: North America, Afr: Africa, S.Am: South America, Oce: Oceanic. Country abbreviations correspond to ISO codes. The symbol (ǂ) represents some uncertainty in the establishment of a species in the EU and thus questions A5 and A6 are completed for these species. Under A6, the percentage corresponds to the estimate area of the EU within the species 95% Kernel. Under questions A8 and A9, uncertainty is represented by (L) low, (M) medium or (H) high.

Species	A.1. Clear taxonomy	A.2. Alien in the EU	A.3. Quality of information sufficient	A.4. Established in the EU	A.5. Invasive outside the EU	A.6. Establishment in the EU	A.7. Spread	A.8. Impact on native plant species	A.9. Impact on ecosystem functions and services	Conclusion of stage 1
Acacia dealbata (Fabaceae)	Yes	Yes (Aus)	High	Yes (ES, FR, IT)			Medium	High (M): forms dense stands displaces native species (Lorenzo et al. 2012)	Medium (L): Nitrogen cycle modifications (Weber 2003)	List IAP
Albizia lebbeck (Fabaceae)	Yes	Yes (Asia)	Low (STOP)	----	----	----	----	----	----	Residual List
Ambrosia confertiflora (Asteraceae)	Yes	Yes (N.Am)	Medium/ High	No	Yes (C.Asia, Oce)	Yes (8.80%)	High	High (M): forms dense stands displaces native species (EPPO 2014a)	Medium (H): Ecosystem modifier (EPPO 2014a)	List IAP
Ambrosia trifida (Asteraceae)	Yes	Yes (N.Am)	Medium/ High	Yesǂ (ES, NL, RO, PL, FR, IT)	Yes (Asia)	Yes (90%)	High	Medium (L): allelopathic and competes with native spp. for nutrients/light (EPPO 2014b)	Low (M): No recorded impacts	Obs List
Andropogon virginicus (Poaceae)	Yes	Yes (N.Am)	High	Yesǂ (FR)	Yes (Asia, N.Am, Oce)	Yes (70.10%)	High	High (H): Allelopathic impacts (Stone 1985)	Medium (H): Promotes fire (Stone 1985)	List IAP
Cardiospermum grandiflorum (Sapindaceae)	Yes	Yes (Afr, S. Am)	Medium	Yesǂ (IT)	Yes (Afr)	Yes (5.10%)	High	High (M): Smothers native spp. (McKay et al. 2010)	Medium (M): Habitat transformer ((Henderson 2001)	List IAP

Species	A.1. Clear taxonomy	A.2. Alien in the EU	A.3. Quality of information sufficient	A.4. Established in the EU	A.5. Invasive outside the EU	A.6. Establishment in the EU	A.7. Spread	A.8. Impact on native plant species	A.9. Impact on ecosystem functions and services	Conclusion of stage 1
Celastrus orbiculatus (Celastraceae)	Yes	Yes (Asia)	High	Yes‡ (GB)	Yes (N.Am, Oce)	Yes (77%)	High	High (H): Suppression native spp. (Fike and Niering 1999)	Medium (H): Negatively affects aesthetics (CABI 2016)	List IAP
Chromolaena odorata (Asteraceae)	Yes	Yes (S.Am)	High	No	Yes (Afr, N.Am, Oce)	No (STOP)	----	----	----	Residual List
Cinnamomum camphora (Lauraceae)	Yes	Yes (Asia)	High	Casual (FR)	Yes (N.Am, Oce)	Yes (35.10%)	High	High (H): Forms monocultures/ Allelopathic impacts (Firth 1979)	Medium (H): Ecosystem modifier (CABI 2016)	List IAP
Clematis terniflora (Ranunculaceae)	Yes	Yes (Asia)	Low (STOP)	----	----	----	----	----	----	Residual List
Cornus sericea (Cornaceae)	No (STOP)	----	----	----	----	----	----	----	----	Residual List
Cortaderia jubata (Poaceae)	Yes	Yes (S. Am)	High	No	Yes (N.Am, Oce)	Yes (55.80%)	High	High (M): Strongly competes for resources (Lambrinos 2000)	High (M): Alters trophic levels/reduces aesthetics (Bossard et al. 2000)	List IAP
Cryptostegia grandiflora (Apocynaceae)	Yes	Yes (Afr)	High	No	Yes (Oce, S.Am)	No (STOP)	----	----	----	Residual List
Egeria densa (Hydrocharitaceae)	Yes	Yes (S. Am)	High	Yes (FR, BE, IT, NL, UK)			High	Medium (H): Displaces native spp. (CABI 2016)	Medium (H): Reduces recreation activities (CABI 2016)	Obs List
Ehrharta calycina (Poaceae)	Yes	Yes (S. Afr)	High	Yes‡ (ES, PT)	Yes (N.Am)	Yes (15.30%)	High	High (M): Outcompetes native plant spp. (Bossard et al. 2000)	Medium (M): Alter fire regimes (Fisher et al. 2006)	List IAP

Species	A.1. Clear taxonomy	A.2. Alien in the EU	A.3. Quality of information sufficient	A.4. Established in the EU	A.5. Invasive outside the EU	A.6. Establishment in the EU	A.7. Spread	A.8. Impact on native plant species	A.9. Impact on ecosystem functions and services	Conclusion of stage 1
Euonymus fortunei (Celastraceae)	Yes	Yes (Asia)	High	Yes‡ (LV)	Yes (N.Am)	Yes (70.10%)	High	High (M): Outcompetes native plant spp. (Bauer and Reynolds 2016)	Medium (H): Ecosystem modifier (Bauer and Reynolds 2016)	List IAP
Euonymus japonicus (Celastraceae)	Yes	Yes (Asia)	Low (STOP)	----	----	----	----	----	----	Residual List
Fallopia baldschuanica (Polygonaceae)	Yes	Yes (Asia)	High	Yes (widespread)			Medium	Medium (M): Smoothers native spp. (EPPO 2012a)	Medium (M): Ecosystem modifier (EPPO 2012a)	Obs List
Gymnocoronis spilanthoides (Asteraceae)	Yes	Yes (N. Am, S.Am.)	High	Yes‡ (HU, IT)	Yes (Asia, Oce)	Yes (14.20%)	High	High (M): forms dense monocultures/ displaces native species (CAB, 2016)	Medium (M): Reduces recreation activities (Weeds CRC 2008, EPPO 2012b)	List IAP
Hakea sericea (Proteaceae)	Yes	Yes (Aus)	Medium	Yes (FR, ES, PT)			High	High (H): Alter composition of native communities (Richardson et al. 1989)	Medium (M): Reduces recreation activities (CABI 2016)	List IAP
Humulus scandens (Cannabaceae)	Yes	Yes (Asia)	High	Yes (FR, IT, HU)			High	High (M): Dense stands outcompete native spp. (EPPO 2007a)	Medium (M): Ecosystem modifier (EPPO 2007a)	List IAP
Hydrilla verticillata (Hydrocharitaceae)	Yes	No (STOP)	----	----	----	----	----	----	----	Residual List

Species	A.1. Clear taxonomy	A.2. Alien in the EU	A.3. Quality of information sufficient	A.4. Established in the EU	A.5. Invasive outside the EU	A.6. Establishment in the EU	A.7. Spread	A.8. Impact on native plant species	A.9. Impact on ecosystem functions and services	Conclusion of stage 1
Hygrophila polysperma (Acanthaceae)	Yes	Yes (Asia)	High	Yes‡ (DE)	Yes (N.Am)	Yes (75.20%)	High	High (H): Dense mats outcompete native plant spp. (Cuda and Sutton 2000)	High (M): Reduces recreation activities (CABI 2016) and blocks drainage systems (Cuda and Sutton 2000)	List IAP
Lespedeza cuneata (Fabaceae)	Yes	Yes (Aus, Asia)	Medium	No	Yes (N.Am, Afr)	Yes (49.10%)	High	High (M): Outcompetes native species/ allelopathic (Coykendall 2011)	Medium (H): Ecosystem modifier (NWCA 2016)	List IAP
Ligustrum sinense (Oleaceae)	Yes	Yes (Asia)	High	Yes‡ (IT, PT, GB)			High	High (M): Reduces abundance and species richness of native plant spp. (Wilcox and Beck 2007)	Medium (H): Ecosystem modifier (Merriam and Feil 2002)	List IAP
Lonicera maackii (Caprifoliaceae)	Yes	Yes (Asia)	Medium	No	Yes (N.Am)	Yes (72.60%)	High	High (M): Reduces plant species richness (Gould and Gorchov 2000; Collier et al. 2002)	Low (H): No recorded impacts	List IAP
Lonicera morrowii (Caprifoliaceae)	Yes	Yes (Asia)	Low (STOP)	----	----	----	----	----	----	Residual List
Lygodium japonicum (Lygodiaceae)	Yes	Yes (Asia)	Medium	No	Yes (N.Am, Oce)	Yes (26.50%)	High	High (H): Dense monocultures outcompete native spp. (Leichty et al. 2011)	Medium (H): Ecosystem modifier (CABI 2016)	List IAP

Species	A.1. Clear taxonomy	A.2. Alien in the EU	A.3. Quality of information sufficient	A.4. Established in the EU	A.5. Invasive outside the EU	A.6. Establishment in the EU	A.7. Spread	A.8. Impact on native plant species	A.9. Impact on ecosystem functions and services	Conclusion of stage 1
Oxalis pes-caprae (Oxalidaceae)	Yes	Yes (Afr)	High	Yes (FR, IT, PT, ES, GB, MT)			Medium	Medium (M): Outcompetes native plant spp. (Petsikos et al. 2007)	Medium (M): Ecosystem modifier (Petsikos et al. 2007)	Obs List
Pennisetum setaceum (Poaceae)	Yes	Yes (Afr, Asia)	High	Yes (IT, PT, ES)			High	High (H): Disrupts primary succession and competes with native species for resources (Cordell and Sandquist 2008)	Low (H): No recorded impacts	List IAP
Pistia stratiotes (Araceae)	Yes	Yes (S.Am)	High	Yes‡ (DE, ES, IT, SI)	Yes (Afr, Asia, N.Am, Oce)	Yes (69.40%)	High	High (M): forms dense mats displaces native species (Hussner 2014; Cilliers et al. (1996)	Medium (H): Reduces recreation activities (Chamier et al. 2012)	List IAP
Prosopis juliflora (Mimosoideae)	Yes	Yes (C.Am, S. Am)	High	No	Yes (Asia, Afr, Oce)	Yes (7.10%)	High	High (M): Outcompetes native plant spp. (Kaur et al. 2012)	High (H): Degrades land/ negative social effect (Choge et al. 2002)	List IAP
Prunus campanulata (Rosaceae)	Yes	Yes (Asia)	Low (STOP)	----	----	----	----	----	----	Residual List
Rubus rosifolius (Rosaceae)	Yes	Yes (Asia)	Low (STOP)	----	----	----	----	----	----	Residual List

Species	A.1. Clear taxonomy	A.2. Alien in the EU	A.3. Quality of information sufficient	A.4. Established in the EU	A.5. Invasive outside the EU	A.6. Establishment in the EU	A.7. Spread	A.8. Impact on native plant species	A.9. Impact on ecosystem functions and services	Conclusion of stage 1
Salvinia molesta (Salviniaceae)	Yes	Yes (S.Am)	High	Yes‡ (IT, PT)	Yes (Afr, N.Am, Oce)	Yes (62.80%)	High	High (M): forms dense monocultures/ displaces native species (Thomas 1981)	High (M): Alters trophic levels, reduces areas for recreation (McFarland et al. 2004; Chilton et al. 2002)	List IAP
Sapium sebiferum (Euphorbiaceae)	Yes	Yes (Asia)	Medium	No	Yes (N.Am, Oce)	Yes (21.70%)	High	High (M): Outcompetes native plant spp. (Camarillo et al. 2015)	High (H): Alters nutrient composition (Bruce et al. 1997)	List IAP
Sphagneticola trilobata (Asteraceae)	Yes	Yes (S.Am)	Medium	Yes‡ (ES, IT)	Yes (Asia, Afr, C. Am, N. Am, Oce)	No (STOP)	----	----	----	Residual List

Table 3. The spread potential scores, uncertainty rating and justification for the 24 plant species assessed under question A7. Spread scores are based on Branquart et al. (2016) where a medium score indicates the species reproduces vigorously vegetatively and/or sexually and spreads mainly in the vicinity of the mother plant; dispersion capacity in the environment rarely exceeds 100–200 m from the mother plant. A high score indicates plant is highly fecund and is regularly observed to spread over distances >500–1000 m from the maternal plant.

Species	Spread score	Uncertainty	Justification
Acacia dealbata	Medium	High	Clonal growth from parental plants. Seed dispersed by birds (DAISIE 2006)
Ambrosia confertiflora	High	Low	Seeds spread over long distances when the hooked spines attach to livestock and wild animals, or can be spread by water, especially during flooding, as the woody burr floats (EPPO 2014)
Ambrosia trifida	High	Medium	Seeds spread naturally via water courses. Seeds also a contaminant of seed stock (CABI 2016)
Andropogon virginicus	High	Medium	Seed spread over long distances by wind (Campbell 1983)
Cardiospermum grandiflorum	High	Medium	Fruit capsules can spread via wind or float along water bodies dispersing the propagules over long distances (EPPO 2012b).
Celastrus orbiculatus	High	Medium	Birds and small mammals spread seed (CABI 2016)
Cinnamomum camphora	High	Medium	Reproduces by seed which are eaten and spread by birds (Firth 1979)
Cortaderia jubata	High	Low	In California, each plant can produce over 100 000 seeds which are wind dispersed (Drewitz and DiTomaso (2004)
Egeria densa	High	Medium	Spread by stem fragments throughout watercourse (State of Washington 2016)
Ehrharta calycina	High	Medium	Seeds spread by wind movement (Wittkuhn 2010)
Euonymus fortunei	High	Medium	Seeds are dispersed by birds and other wildlife and by water (NPWG 2010)
Fallopia baldschuanica	Medium	Low	Spread by seed, vegetatively and rhizomes (EPPO 2007)
Gymnocoronis spilanthoides	High	Medium	Broken stem fragments are spread by water currents, and can also be accidentally spread by machinery (e. g. boats, trailers, etc.) or animal hooves, and grow into a new plant when settling in a stream bed, and then form new colonies (EPPO 2012b).
Hakea sericea	High	Low	Winged seeds which are produced in large numbers are dispersed by wind (Richardson et al. 1987)
Humulus scandens	High	Low	Reproduces by seeds which are spread by wind and water (EPPO 2007a)
Hygrophila polysperma	High	Low	Brittle stem fragments are capable of spreading by water currents (Kasselmann 1994). Spread can be facilitated by recreational activities (DCR 2003).
Lespedeza cuneata	High	Medium	Aggressively spreading species. Reproduces by seed as well as vegetatively (Bugwood 2016)
Ligustrum sinense	High	Low	Prolific seed producer and the fruit is spread by birds up to 1 km from parental plant (Swarbrick et al. 1999).
Lonicera maackii	High	Low	Birds and mammals disperse seeds over long distances in the USA (Castellano and Gorchov 2013).

Species	Spread score	Uncertainty	Justification
Lygodium japonicum	High	Low	Tiny spores are readily dispersed by wind (CABI 2016)
Oxalis pes-caprae	Medium	Medium	Vegetative reproduction dispersed by agricultural activity and water (DAISIE 2006).
Pennisetum setaceum	High	Low	Seeds spread over long distances by wind (PCA 2005)
Pistia stratiotes	High	Low	Long dispersal of plants facilitated by water movement (Hussner and Heiligtag 2013). Additional spread likely from water birds and recreational activities
Prosopis juliflora	High	Low	Seed is spread by birds and mammals over long distances. Seeds can become incorporated into waterbodies facilitating spread (CRC Weed Management 2003)
Salvinia molesta	High	Medium	The floating form of the plant facilitates its spread within waterbodies (McFarland et al. 2004); likewise, flooding also has the potential to carry plants to new waterbodies or wetland habitats (McFarland et al. 2004).
Sapium sebiferum	High	Medium	Seeds can become incorporated into waterbodies and disperse through the system. Birds eat and disperse seeds (Jubinsky and Anderson 1996)

plant species. Four species, *Ambrosia trifida* L., *Egeria densa* Planch., *Fallopia baldschuanica* (Regel) Holub and *Oxalis pes-caprae* L., were assigned to the EU Observation list due a medium impact score.

Stage 2 (Risk management)

Of the 22 species assessed under stage 2, 19 were considered as a high priority for a PRA at the EU level (Table 4). All species were regarded as having the potential for further spread in climatically suitable regions (see Table 4).

Andropogon virginicus L., *Humulus scandens* (Lour.) Merr., and *Lespedeza cuneata* (Dum. Cours.) G.Don, were regarded as having the highest potential for further spread where each could colonise 4 biogeographical regions.

Three species, *Euonymus fortunei* (Turcz.) Hand.-Maz., *Ligustrum sinense* Lour. and *Lonicera maackii* (Rupr.) Maxim., were considered low priority for PRA as all are widely cultivated within the EU without showing significant signs of invasive behaviour (Table 4). However, all three species are known to be invasive in North America, particularly in the eastern States which have similar climatic zones to regions in Europe (see Köppen-Geiger climate classification, Kottek et al. 2006). Based on the precautionary principle, national measures could be applied to these species, including country specific PRA.

Most species (68 %) evaluated under question B3 (can the risk of introduction and spread into and within the EU be effectively controlled by trade restrictions?), are sold within the EU and therefore a European level PRA would be required to assess if trade restrictions could prevent further introduction and spread (Table 3). Where trade restrictions were regarded as ineffective, as in the case of *Ambrosia confertiflora* DC, *Andropogon virginicus*, *Cortaderia jubata* and *Prosopis juliflora* (Sw.) DC., members of the workshop considered that cost-effective integrated control actions could be applied against these species and therefore they are a priority for a European level PRA.

Discussion

Globally, numerous prioritisation schemes have been specifically designed to address specific taxonomic groups (Brunel et al. 2010, Worner et al. 2013), regions or habitats (Dawson et al. 2015), pathways (NOBANIS 2015) or requirements of specific regulations (see McGeoch et al. 2016). With the implementation of the IAS Regulation, the European Commission has placed a clear focus on mitigating the negative impacts of IAS on biological diversity and ecosystem services, coupled with an underlying requirement to focus efforts on prevention rather than cure. Often, risk management components are lacking in prioritisation schemes (Heikkilä 2011), even though there is a clear advantage of incorporating such aspects to prioritise species that can be effectively controlled over other more difficult species (Hulme 2009). The current EU

Table 4. Results from the prioritisation exercise (Stage 2: risk management). Based on the potential for further spread and available prevention and control methods, 19 species where identified a priority for an EU level risk assessment. Under question B1, the potential biogeographical regions that could be invaded are listed in brackets where abbreviations follow: ATL: Atlantic, CON: Continental, MED: Mediterranean, STE: Steppic. Under question B2 countries (abbreviations correspond to ISO codes) are indicated where the species has shown evidence of invasiveness.

Species	B.1. Potential for further spread	B.2. Widely cultivated without invasive behaviour	B.3. Can intro/ spread be reduced by trade restrictions	B.4. Can intro/spread be reduced by other preventative action	B.5. Can populations be cost-effectively eradicated	Conclusion of stage 2
Acacia dealbata	Yes (ATL, MED)	No (FR, PT)	Yes (forestry spp.)			Priority for EU RA
Ambrosia confertiflora	Yes (MED)	No (not established)	No (seed contaminant)	No (small seed difficult to detect)	Yes (effective management measures exist and risk management can identify national or international measures)	Priority for EU RA
Andropogon virginicus	Yes (ATL, CON, MED, STE)	No (FR)	Yes* (internet sale)	No (small seed difficult to detect)	Yes (large conspicuous grass, with existing management methods)	Priority for EU RA
Cardiospermum grandiflorum	Yes (MED)	No (Not widely planted, Inv. similar climatic regions)	Yes (traded)			Priority for EU RA
Celastrus orbiculatus	Yes (ATL, CON, MED)	No (Not widely planted, inv. similar climatic regions)	Yes (traded)			Priority for EU RA
Cinnamomum camphora	Yes (MED)	No (Not widely planted, inv. similar climatic regions)	Yes (traded)			Priority for EU RA
Cortaderia jubata	Yes (ATL, MED)	No (not established)	Yes* (internet sale)	No (small seed difficult to detect)	Yes (large conspicuous grass, with existing management methods)	Priority for EU RA
Ehrharta calycina	Yes (MED)	No (Not widely planted, inv. similar climatic regions)	Yes (traded)			Priority for EU RA

Species	B.1. Potential for further spread	B.2. Widely cultivated without invasive behaviour	B.3. Can intro/ spread be reduced by trade restrictions	B.4. Can intro/spread be reduced by other preventative action	B.5. Can populations be cost-effectively eradicated	Conclusion of stage 2
Euonymus fortunei	Yes (CON, STE)	Yes (STOP)	----	----	----	Not a priority for RA (national measures)
Gymnocoronis spilanthoides	Yes (MED)	No (HU, IT)	Yes (widely sold within EU)			Priority for EU RA
Hakea sericea	Yes (ATL, MED)	No (PT)	Yes (traded)			Priority for EU RA
Humulus scandens	Yes (ATL, CON, MED, STE)	No (FR)	Yes (traded)			Priority for EU RA
Hygrophila polysperma	Yes (MED)	No (DE)	Yes (traded)			Priority for EU RA
Lespedeza cuneata	Yes (ATL, CON, MED, STE)	No (not established)	Yes (currently absent from EU)			Priority for EU RA
Ligustrum sinense	Yes (ALT, CON, MED)	Yes (STOP)	----	----	----	Not a priority for RA (national measures)
Lonicera maackii	Yes (ALT, CON, MED)	Yes (STOP)	----	----	----	Not a priority for RA (national measures)
Lygodium japonicum	Yes (MED)	No (not established)	Yes (currently absent from EU)			Priority for EU RA
Pennisetum setaceum	Yes (ATL, MED)	No (ES)	Yes (traded)			Priority for EU RA
Pistia stratiotes	Yes (MED)	No (DE)	Yes (widely traded within EU)			Priority for EU RA
Prosopis juliflora	Yes (MED)	No (not established)	Yes* (internet sale)	No	Yes (large conspicuous shrub species, with existing management methods)	Priority for EU RA
Salvinia molesta	Yes (MED)	No (AU, BE, FR, IT, PT)	Yes (widely traded within EU)			Priority for EU RA
Sapium sebiferum	Yes (ATL, MED)	No (not established)	Yes (currently absent from EU)			Priority for EU RA

prioritisation scheme has been specifically designed to incorporate the requirements of the IAS Regulation and to the best of our knowledge, this is the first-time invasive alien plant species have been prioritised using a scheme compliant with the IAS Regulation.

This study identified 19 globally invasive alien plant species with a high priority for a PRA at the EU level. As shown in our results, all 19 species comply with the IAS definition and criteria of art. 4 of the IAS Regulation, i.e. alien species being capable of causing major detrimental impacts to biodiversity and the associated ecosystem services after establishment and spread within the territory of the EU. Within the first stage of the prioritisation scheme, four species (*A. trifida*, *E. densa*, *F. baldschuanica* and *O. pes-caprae*) were assigned to the EU Observation List highlighting that at the current time the species are likely to cause only a moderate detrimental impact to biodiversity and associated ecosystem services. *A. trifida* has become a major weed of annual crops in the US (Weaver 2003) and is a threat for the economy where it has established in Europe, especially in SW France (Chauvel et al. 2015), while *O. pes-caprae* has impacts on livestock. It should be noted that the placement of species in the three lists is not a definitive placement and each species should be reviewed as and when new information comes to light. This is particularly important for any species included in the EU list of Minor Concern and EU Observational List.

In the first stage of the prioritisation scheme, eleven species were assigned to the Residual List and thus did not qualify for further assessment. Having a clear understanding of the taxonomic identity of a species is an essential component in any prioritisation, and subsequent PRA. This is important to ensure that the assessment is performed on a distinct organism (IPPC 2016) but also to ensure that information used in the assessment is relevant to the organism under consideration (Elith et al. 2012). In our initial list of 37 species, the taxonomy of one species, *Cornus sericea*, was identified as being uncertain as in Europe naturalised plants belong to a complex of hybrids of *C. sericea* and *C. alba* (Q-Bank 2016). It has also been suggested that *C. alba* is conspecific with *C. sericea* (National Botanic Garden of Belgium 2016). We suggest that further research is carried out on the exact identity of the species within Europe before any PRA is conducted to reduce uncertainty.

If an invasive plant is native to part of the European Union, this would preclude its inclusion on the list of species of Union concern. Although *Hydrilla verticillata* is often considered non-native to Europe, there is some uncertainty to the status of the species and Lansdown (2013) details the species as native to Belarus, Ireland, the United Kingdom (southern Scotland) and the Russian Federation. There is additional uncertainty of its native status in Latvia and Poland (Cook and Lüönd 1982). In the absence of a pan-global biogeographical molecular study the uncertainty of native populations within Europe will remain (Zhu et al. 2015). It should be noted that provisions are made within the IAS Regulation (Article 11) for species native to the Union, where their inclusion on national lists can be used to enhance regional cooperation.

Most species included in the Residual List (75 %) warrant their place due to the lack of current information on the species. A PRA is only as robust as the scientific information which is used to compile the assessment and even though uncertainty ratings can go some way to capturing data gaps, or conflicting information, without some baseline data consideration is needed to whether a PRA is warranted. Based on the lack of quantitative impact studies, and to some extent information on the biology and ecology of the species (at a global scale), we considered *Albizia lebbeck*, *Clematis terniflora*, *Euonymus japonicus*, *Lonicera morrowii*, *Prunus campanulata* and *Rubus rosifolius* are not suited for a PRA at this time. We do suggest that a comprehensive literature review is conducted periodically for each species in the Residual List, including those species where there is uncertainty in potential for establishment (*C. odorata*, *C. grandiflora* and *S. trilobata*). If new scientific information comes to light that may change the outcome of the prioritisation, the species should be re-evaluated.

Impact studies can be biased to species which are widespread and/or high-profile species to particular sectors of society (Hulme et al. 2013). When considering species which are absent from the EU, there is a clear need to use the invasion history from another region as a proxy (Gallardo et al. 2015). As already mentioned, most species assessed under question A9, impacts on ecosystem functions and services, received a medium score (69 %) with a high level of uncertainty (55 %). This is in contrast with the previous question on impacts on native species where 84 % of the species received a high score with medium uncertainty (68 %). This is not a surprise as impacts can be ambiguous to define in relation to ecosystem services and impacts can be inconspicuous in many studies conducted over a short timeframe (Eviner et al. 2012). It is however fair to note that our understanding of the effects of invasive plants on ecosystem services is growing (Vilà et al. 2010), and with the prominence of ecosystem services in the IAS Regulation further studies will undoubtedly follow.

All 22 species evaluated under stage 2 have potential for further spread, though three species, namely *Euonymus fortunei*, *Ligustrum sinense* and *Lonicera maackii* were not considered a priority for an European level PRA due to being widely cultivated within the region without showing any signs of invasive behaviour. *E. fortunei* has been cultivated within the region since the late 1800s where it is grown in parks and gardens (Personal Communication, John David, Royal Horticultural Society, UK, 2016). It has however, been identified in the eastern USA as a species spreading into native plant communities (Missouri Botanical Garden 2016), and research has shown it causes native species decline (Bauer and Reynolds 2016, Mattingly et al. 2016). We recommend that Member States monitor these species, e.g. considering the possibility to join a network of sentinel gardens (to detect as soon as possible any sign of potential invasiveness) (Visser et al. 2014).

In the prioritisation scheme, questions B3-B5 focus on the cost effectiveness of prevention and management measures and assess if the introduction and spread of the species can be reduced by trade restrictions, other preventative actions (pathway management) or cost-effective management in the field (Branquart et al. 2016). A positive answer to any of the three questions indicates that a full PRA may identify actions to mitigate entry or spread. The risk from the majority of species (84 %) could be mitigated with trade restrictions as most are either traded or absent from the EU. However, for *A. confertiflora*, *A. virginicus*, and *C. jubata* it was considered that pathway management would be ineffective in detecting and preventing the incurrence of small plant propagules. As all species are relatively large in form, management *in situ* should be a feasible cost-effective option for isolated incurrences, particularly as management options exist for each species (see Panetta 2015).

It should be noted for *Andropogon virginicus*, *Prosopis juliflora* and *Cortaderia jubata*, trade in these species is predominately via internet sites such as eBay and Amazon and as such any trade restrictions may be ineffective in the absence of greater enforcement of existing regulations (Lenda et al. 2014). The volume of movement of these species is likely to be low along this pathway, but not necessarily insignificant, as has been shown for other species where a small number of introductions have resulted in invasive populations (for example *Polygonum perfoliatum* in the USA (IPANE 2016)).

The EU prioritisation scheme does not consider potential impacts which may be realised because of climate change scenarios, or indeed the potential for further spread and establishment because of future climatic projections. Of course, the effect of climate change on a species is a key consideration in any subsequent PRA but we suggest that the detailed analysis needed to address this issue is not suited to a prioritisation scheme. We reiterate the point made in Branquart et al. (2016) that a prioritisation scheme is no substitution for a comprehensive PRA but instead acts as a valuable tool to filter out those species where a PRA is not currently needed allowing efforts to focus on those species where a detailed RA is required.

In conclusion, in utilising the EU prioritisation process for alien plants, 19 species have been identified as high priorities for PRA. These species present a prominent risk to the EU, either now or in the future and thus warrant a full PRA. Whether these species are eventually included on the list of Union concern remains to be seen and will depend, in part, on the outcome of the subsequent PRA and decision makers of the Member States.

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