Invasive alien plants most harmful to pollinators in the EU

and their management focusing on non-chemical methods

This document shall be cited as:

Nunes AL (ed.), Pergl J, Rabitsch W (2024). Invasive alien plants most harmful to pollinators in the EU and their management focusing on non-chemical methods. Technical note prepared by IUCN for the European Commission.

Reviewers: Giuseppe Brundu, Katie Costello, Helen Roy, Kevin Smith, Tamryn Venter

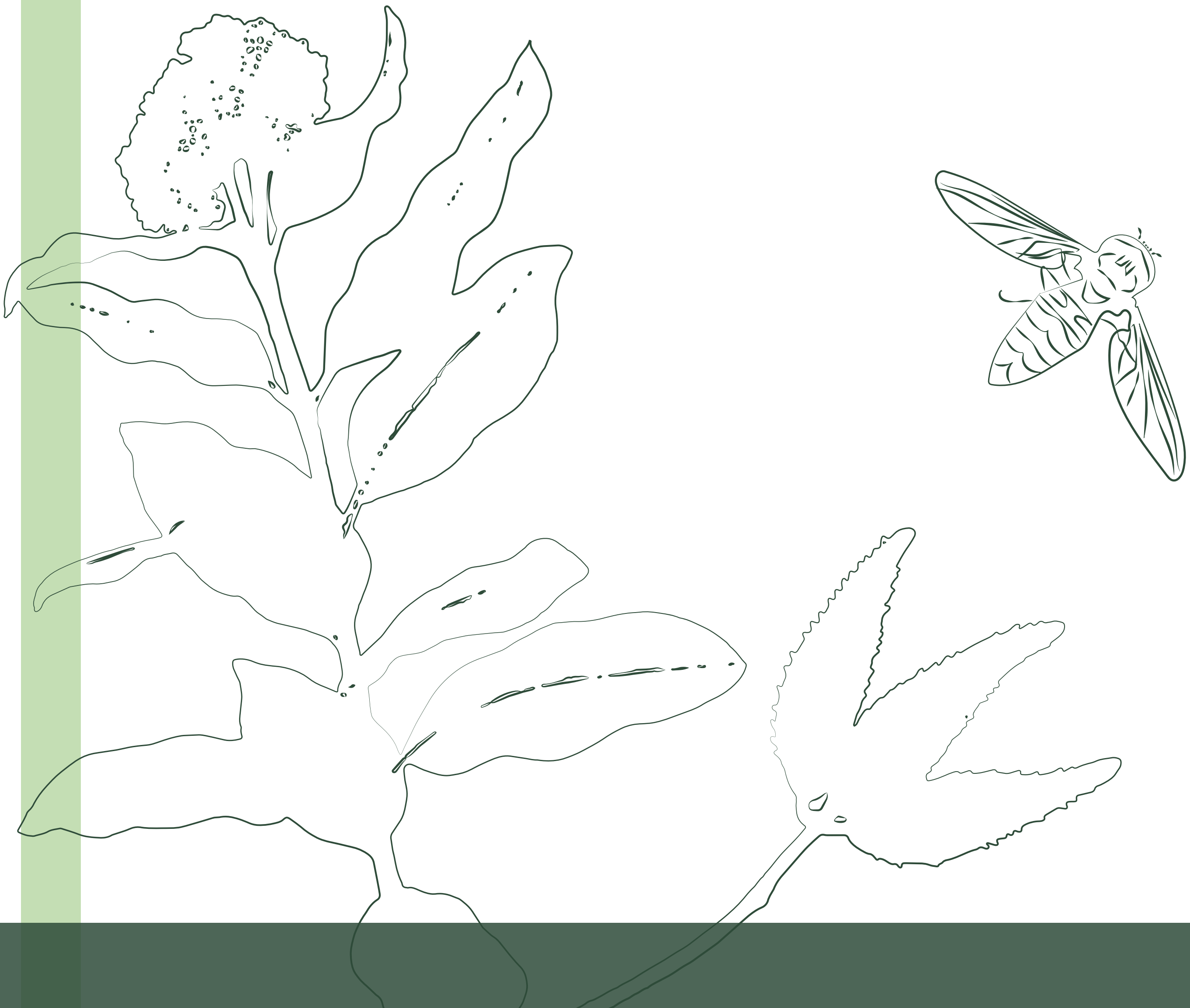
Date of completion: January 2024

Front Cover: European peacock butterfly (*Aglais io*) © Mathias Appel, (CC0 1.0) via Flickr

Design and layout: Sabina Salussolia

Comments which could support improvement of this document are welcome. Please send your comments by e-mail to ENV-IAS@ec.europa.eu.

Contents



1. Summary

This brief report summarises information on terrestrial invasive alien plant species (IAPS) in the EU having a negative impact on pollinators, with a focus on the non-chemical methods that can be used to manage them. A limited systematic literature search revealed 52 publications on IAPS affecting pollinators since 2000 and highlighted 35 relevant IAPS. These plants were grouped and ranked using a combination of their current distribution in Europe and their impact on pollination. For a selection of these IAPS, information on their native and alien range, their impact on pollinators, and the non-chemical management methods that can be used to tackle them, are provided. Some of the species in focus are listed as species of Union concern under Regulation (EU) No 1143/2014. Based on the number of publications demonstrating negative impacts on pollinators, the potentially most harmful – and not yet regulated – species are two goldenrods, *Solidago canadensis* and *Solidago gigantea.*

The management of IAPS is key for conserving ecosystems. Classical physical methods of control, such as mechanical removal and manual uprooting, are effective and can be used for some IAPS in specific conditions. However, it is important to note that, for many others, these methods have had limited efficacy, therefore chemical methods having become an important tool in effective IAPS management. Some chemical treatments can, of course, lead to negative impacts upon the wider environment, but non-target effects can be limited by following recommended guidance for implementation. Indeed, chemical control may sometimes have less non-target effects than repeated physical interventions (as they reduce the need for frequent site visits and disturbance to the ecosystem). At the same time, there is growing interest in biological control (biocontrol) methods, which involve introducing herbivores or pathogens to target IAPS. Biocontrol can offer a promising long-term solution to suppress IAPS, while minimising the use of synthetic chemicals or disturbance caused by physical management. However, it is essential to carefully assess the potential risks and impacts of biocontrol through a thorough evidence-based risk assessment before implementation, as well as to continually monitor its effectiveness.

Canadian goldenrod (*Solidago canadensis*) © Forest and Kim Starr, (CC BY 2.0) via Flickr

Silverleaf nightshade (*Solanum elaeagnifolium*) © CarlsbadCavernsNPS, (PDM 1.0) via Flickr

2. Introduction

IAPS can have direct and indirect impacts on biodiversity. They can outcompete native plants, e.g. by outgrowing and overshadowing or by releasing growth-inhibiting compounds (allelopathic effects), and modify habitats, e.g. by nitrogen-fixation and modification of the soil chemistry (e.g. Liao et al. 2007, Gaertner et al. 2009, Hejda et al. 2009, Vilà et al. 2011, Pyšek et al. 2012). Positive, neutral and negative impacts on plant-pollinator interactions and networks can be found (see e.g. Bjerknes et al. 2007, Morales & Traveset 2009, Ferrero et al. 2013, Stout & Tiedeken 2016, Vanbergen et al. 2018 for an overview), but there is a lack of detailed understanding on how IAPS influence habitats, and subsequently the complex and context-dependent trophic and functional interactions between species (e.g. Albrecht et al. 2014, Bezemer et al. 2014, Litt et al. 2014, Arceo-Gómez & Ashman 2016, Charlebois & Sargent 2017). Recently, Johnson et al. (2022) found that competition for pollinators destabilised plant interactions and disrupted plant coexistence.

The negative impacts of IAPS on pollinators and pollination can be direct, e.g. the IAPS' pollen or nectar can be toxic to pollinators (Adler 2001, Stevenson et al. 2017, Rivest & Forrest 2019), and indirect, e.g. by replacing native plants that are visited by specialist pollinators or by promoting generalist pollinators that then may outcompete specialist pollinators (e.g. Traveset & Richardson 2006, Valtonen et al. 2006, Moroń et al. 2009, Stout & Morales 2009, Schweiger et al. 2010, Hanula & Horn 2011a, 2011b, Hudson et al. 2013, Scheper et al. 2014, Fenesi et al. 2015, Stout & Tiedeken 2016, Parra-Tabla & Arceo-Gómez 2021, Zaninotto et al. 2023). Additionally, IAPS with an ornamental history often have showy flowers, which attract pollinators, and flower late in the season, which may affect plant-pollinator interactions. Little known, but with potentially severe consequences, is the role of IAPS as hubs for pathogen transfer and facilitation of pathogen spread (e.g. Proesmans et al. 2021, Najberek et al. 2023).

Many pollinator species are in decline and becoming increasingly threatened (e.g. Van Swaay et al. 2010, Nieto et al. 2014, Vujić et al. 2022) and so, in 2018, the European Commission adopted the first-ever EU framework to address the decline of wild pollinators – the EU Pollinators Initiative. Action 8 of the EU Pollinators Initiative aims to reduce the impacts of Invasive alien species (IAS) on pollinators (e.g. COM 2021). In line with this, and under the EC funded project "Technical support related to the implementation of the EU Pollinators Initiative", IUCN (2020) provided a summary of the possible impacts of IAS on pollinators, emphasizing that native wild pollinators are potentially affected by ecosystem modification, competition, hybridisation, predation and disease transmission and parasites. While the latter three mechanisms mostly refer to the impacts caused by animals (e.g. *Vespa velutina*, *Megachile sculpturalis*, *Harmonia axyridis*, commercial honeybees and bumblebees), the modification of terrestrial habitats mostly refers to IAPS. The document mentions that "*When invasive alien plants dominate an area they transform the availability of nectar and pollen, often from a diverse suite of floral species that may provide nutrition at different times of year. While the invasive plant may provide nutrition, it may only do so for a certain part of the year and this may only favour certain pollinator groups or species, usually those with generalist feeding behaviours, negatively affecting those specialised pollinator species.*" (IUCN 2020).

It must be mentioned, however, that there is not a single determining factor explaining the decline of pollinators, and that many factors need consideration in attempts to support and promote pollinators' conservation (IPBES 2017). Van Swaay et al. (2010) analysed the major threats to European butterflies and found that agricultural intensification, abandonment, climate change (incl. droughts) and change of woodland management are leading causes. Invasive alien species were found to threaten endemic species, especially on islands, e.g. the introduction of the alien braconid parasitoid *Cotesia glomerata* might have contributed to the extinction of the Madeiran Large White *Pieris wollastoni* (Wiemers et al. 2022) and the decline of the Canary Islands Large White *Pieris cheiranthi* (Lozan et al. 2008). Nieto et al. (2014) analysed the major threats to European bees and found that agricultural expansion and intensification, livestock farming and ranching, pollu-

Canary Islands Large white (*Pieris cheiranthi*) © Quartl, (CC BY 3.0) via Wikimedia Commons

A Marmalade hoverfly (*Episyrphus balteatus*) feeding on the nectar of Common asphodel (*Asphodelus aestivus*) flowers © Zeynel Cebeci, (CC BY-SA 4.0) via Wikimedia Commons

tion and residential and commercial development are leading drivers of extinction risk to bees. The importance of IAS is recognised, but due to the lack of data and complex interactions with other factors, it is not explicitly addressed. Vujić et al. (2022) analysed the major threats to European hoverflies and found that agriculture and aquaculture, residential and commercial development, natural system modifications (e.g. fire, dams), and climate change and severe weather, are leading causes. Invasive and other problematic species affect 52 species of European hoverflies (11 of which are threatened), mostly due to predation and competition from the Harlequin ladybird (*Harmonia axyridis*) (e.g. Alhmedi et al. 2010).

The revised EU Pollinators Initiative, which follows up on the review of progress in implementing the EU Pollinators Initiative and builds on comprehensive stakeholder consultations and institutional feedback, sets out actions to be taken by the EU and its Member States to reverse the decline of pollinators by 2030 (COM 2023). These actions are reflected under three priorities, one of which is 'PRIORITY II: Improving pollinator conservation and tackling the causes of their decline'. Under Priority II, again Action 8 aims to 'Reduce the impacts of invasive alien species on pollinators' through three specific actions. Action 8.1. aims to assess threats to pollinators from IAS not yet included in the list of IAS of Union concern under Regulation (EU) No 1143/2014 and preparing risk assessments for the most problematic ones. Action 8.2. seeks to assess management options for IAPS most harmful to wild pollinators, with a view to increasing the availability, uptake, and effectiveness of non-chemical management options. Finally, Action 8.3 commits to developing guidelines to promote the use of pollinator-friendly native plants and seed mixes in areas including private gardens, public areas, farmland, and forests.

The aim of this document is to partially address Actions 8.1. and 8.2., by identifying the IAPS that seem the most harmful to pollinators in the EU, at the same time outlining options for their management, with a focus on the use of non-chemical methods.

3. Terrestrial invasive alien plant species currently most harmful to pollinators in the EU

3.1. Methods

3.1.1. Species selection

Within the consultation process of the EU Pollinators Initiative, thematic workshops on the most serious threats were organised, including one on "Measures to tackle pressures from invasive alien species on wild pollinators" (Barov et al. 2022). The workshop report lists 20 IAPS, most of them also mentioned by the IUCN report on managing IAS to protect wild pollinators (IUCN 2020). Searching for spatial patterns, Rabitsch et al. (2021) analysed distributional data of three species within the EU (*Asclepias syriaca*, *Heracleum mantegazzianum*, *Impatiens glandulifera*), and concluded that currently available data do not allow for defining potential areas at risk to pollinators from these IAPS.

The IAPS included in IUCN (2020), Rabitsch et al. (2021) and Barov et al. (2022) served as a starting point for the current study. A partial literature research was performed on 27.7.2023 using Google Scholar to identify additional IAPS potentially having negative impacts on native pollinators. The search string "invasive plant species pollination" resulted in more than 68,000 hits, and more than 18,000 after filtering for publications between the years 2000 to 2023. The titles of the first 200 publications were screened for additional IAPS. Additionally, publications referring to plants having pollen or nectar toxic for pollinators according to Adler (2001) and Stevenson et al. (2017), and alien to Europe, were added. This resulted in a list of 35 IAPS potentially having negative impacts on native pollinators, which are listed and described in Table 2 below. Finally, a Google search using the string "scientific name of the IAPS"+"invasive" and "scientific name of the IAPS"+"pollination" was performed, searching for published impacts on pollination. In total, 52 publications were found and read in full. In not few cases, however, only anecdotic evidence of impacts of IAPS on pollinators was found. Repositories such as the CABI Invasive Species Compendium[1], the Global Invasive Species Database[2] and the EPPO Global Database (EPPO 2023) were consulted for additional information.

Publications of European species being alien elsewhere within Europe have been excluded (e.g. the Purple loosestrife *Lythrum salicaria;* Flanagan et al. (2010)), except for *Rhododendron ponticum*. IAPS which are not yet present in the wild in Europe were also not considered (e.g. the Amur honeysuckle *Lonicera maackii*, native to temperate Asia and invasive in North America; McKinney & Goodell (2010), although IAPS currently present in cultivation only, can have negative impacts as well. This latter species, however, could be considered in a future horizon scanning of IAPS in Europe. Also excluded are IAPS known to occur in Europe, but where the impact has been analysed outside of Europe. Examples of this are *Acacia saligna*, a species of Union concern, native to Australia and present in the Mediterranean region, causing reduced flower visitation in a native plant in South Africa (Gibson et al. 2013) and *Vincetoxicum rossicum*, native to Ukraine and Russia and introduced to North America and Europe, negatively affecting the migratory monarch butterfly (*Danaus plexippus)*, being toxic to its larvae in North America (DiTommaso & Losey 2003, Ernst & Cappuccino 2005).

Giant hogweed (*Heracleum mantegazzianum*) © antefixus21, (CC BY-NC-ND 2.0) via Flickr

[1]https://www.cabidigitallibrary.org/product/qi
[2]http://www.iucngisd.org/gisd/

3.1.2. Species prioritisation

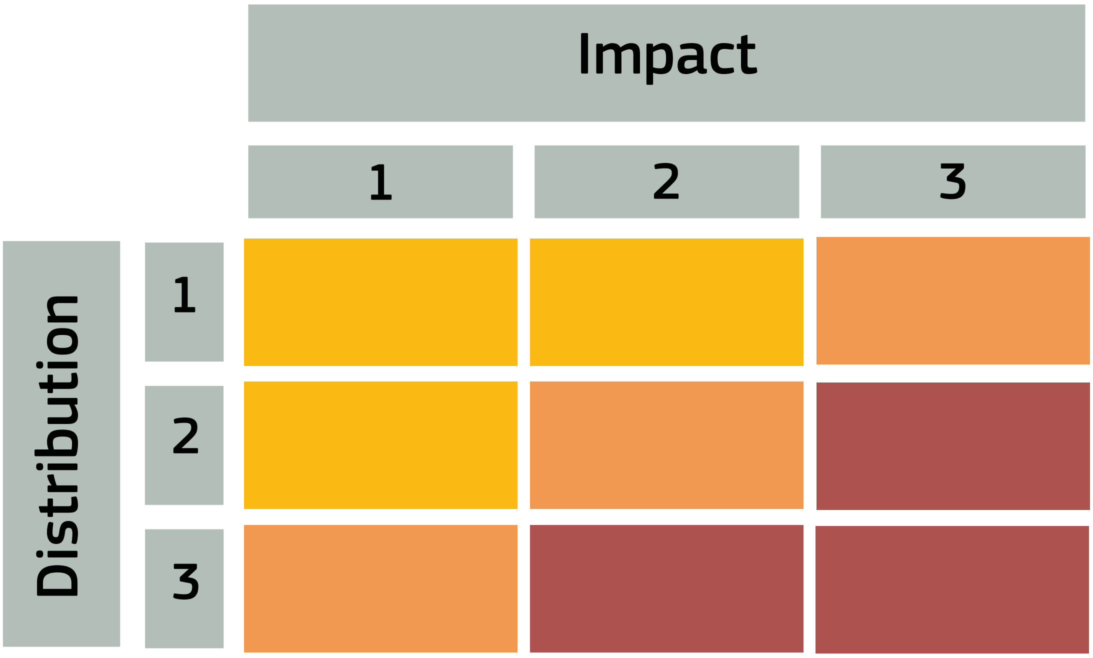
Impact on native pollinators and on pollination was assessed on a three-point scale: 1 = no published evidence of impact found (although publications mentioning potential impacts exist), 2 = limited evidence of impact (one published paper supporting negative impacts on pollinators in Europe), 3 = high evidence of impact (more than one published paper supporting negative impacts on pollinators).

Distribution of the species in Europe was assessed on a three-point scale, based on the number of occurrence records for Europe in GBIF (assessed 11.01.2024; applied filters: Basis of records: Human observation; Continent: Europe): 1 = < 10,000 records, 2 = 10,000 to 100,000 records and 3 = > 100,000 records.

Based on the combination of these two criteria, a species matrix was created (Table 1). Combinations with many occurrence records and high evidence of impacts are highlighted in red, the opposite is highlighted in yellow and intermediate scores are indicated in orange.

For the ten species (or groups of species) highlighted in red, brief summaries are provided in different species factsheets in Section 5, including information on native and alien range, a description of the impacts on pollination, a brief account of their biology and ecology, and relevant management methods. In addition, this information is provided for two species with a currently small distribution (score 1), but potentially high impact (score 3) on pollination in Europe.

Table 1: Species matrix with the possible combinations of the three-point scales for Impact (on pollination) and Distribution (within Europe) of invasive alien plant species.



3.2. Results

Table 2 presents the results of the IAPS mentioned in the analysed references, retrieved through the species selection and prioritisation described above.

Table 2: Alphabetic list of the 35 IAPS (invasive alien plant species) with the potential to negatively affect native plants, pollinators and pollination. D = Distribution criterion (1 = < 10,000 records, 2 = 10,000 to 100,000 records, 3 = > 100,000 records); I = Impact criterion (1 = no evidence, 2 = limited evidence, 3 = high evidence); * = reference not found in the partial literature search and added later; [i] = species of Union concern. A brief summary is provided for each species here, except for those species most impactful to pollinators in the EU (marked in bold), for which a more extensive description is provided in Section 5.

Species	Common name	Native range	# European records (GBIF)	D	Impact	I	Source
Amorpha fruticosa	False indigobush	North America	9,720	1	Outcompeting native plants; Habitat modification; Pollinator competition	1	Barov et al. (2022)
Anecdotic sources mention that the showy and honey-rich flowers attract pollinators and therefore are in competition with native plant species for pollination. However, no published evidence for such an impact could be found. It is considered sometimes as a valuable honey plant.							
Araujia sericifera	Moth plant	South America	11,455	2	Outcompeting native plants; Habitat modification; "Mothcatching"	1	Coombs & Peter (2010), Barov et al. (2022)
The species is considered invasive because of its ability to outcompete and replace native plants, as well as to modify vegetation structure and natural succession. In Europe, it is present in ES, FR, GR, IT, and PT. It is famous for "catching" Lepidoptera and other insect flower visitors if their mouthparts become wedged within the flowers, but it is not a carnivorous plant. Bumblebees and scoliid wasps are pollinators of the plant in Europe. There is no evidence of a significant negative effect on pollinators at the population or community level in Europe.							
Arundo donax	Giant reed	Temp. and Trop. Asia	45,720	2	Outcompeting native plants; Habitat modification	1	Barov et al. (2022)
Invasive in riparian habitats and considered one of the "100 worst" according to IUCN ISSG[3]. It is a transformer species which increases fire susceptibility, water use and erosion. Negative effects on ground arthropod invertebrates are documented (e.g. Herrera & Dudley 2003, Maceda-Veiga et al. 2016), but no evidence for significant negative impacts on pollinators could be found in Europe or elsewhere (see e.g. CABI 2014a, Jiménez-Ruiz et al. 2021).							
***Asclepias syriaca*[i]**	**Common milkweed**	North America	5,338	1	Outcompeting native plants; Habitat modification	3	Bagi (2008)*, Kelemen et al. (2016)*, Szigeti et al. (2020)*, Rabitsch et al. (2021), Barov et al. (2022), Kovács-Hostyánszki et al. (2022)*
Baccharis halimifolia[i]	Groundsel-bush	North America	8,703	1	Outcompeting native plants; Habitat modification	1	Barov et al. (2022)
Invasive in saltmarsh and sand dune habitats in western and southern Europe (BE, ES, FR, IT). Monodominant stands replace native vegetation and reduce phytophagous insect diversity (e.g. Mallard 2008), but no evidence for significant negative impacts on pollinators could be found in Europe or elsewhere (e.g. Fried et al. 2016, CABI 2019).							

[3] https://www.iucngisd.org/gisd/100_worst.php

<table>
<tr><th>Species</th><th>Common name</th><th>Native range</th><th># European records (GBIF)</th><th>D</th><th>Impact</th><th>I</th><th>Source</th></tr>
<tr><td>Buddleja davidii</td><td>Butterfly bush</td><td>Temp. Asia</td><td>185,423</td><td>3</td><td>Outcompeting native plants; Habitat modification</td><td>3</td><td>Giuliano et al. (2004)*, Tallent-Halsell & Watt (2009)*, Corcos et al. (2020)*, Barov et al. (2022)</td></tr>
<tr><td>Carpobrotus edulis, C. acinaciformis and their hybrids</td><td>Ice plant</td><td>South Africa</td><td>18, 535 (C. edulis)
1,467 (C. acinaciformis)</td><td>2</td><td>Outcompeting native plants; Habitat modification</td><td>3</td><td>Moragues & Traveset (2005), Suehs et al. (2005)*, Bartomeus et al. (2008b), Jakobsson et al. (2008), Vilà et al. (2009), Barov et al. (2022), León et al. (2023)*</td></tr>
<tr><td>Catalpa speciosa</td><td>Northern Catalpa</td><td>North America</td><td>25</td><td>1</td><td>Toxic (iridoid glyco-sides deters ants and butterflies)</td><td>1</td><td>Stephenson (1981, 1982)</td></tr>
<tr><td colspan="8">Introduced to Asia and Europe as an ornamental tree for urban gardens and parks. It is pollinated by bees during the day and by moths in the night. No evidence of significant negative impacts on pollinators could be found in Europe or elsewhere. It is considered sometimes as a valuable honey plant.</td></tr>
<tr><td>Cortaderia selloana</td><td>Pampas grass</td><td>South America</td><td>19,033</td><td>2</td><td>Outcompeting native plants; Habitat modification</td><td>1</td><td>Barov et al. (2022)</td></tr>
<tr><td colspan="8">Mentioned by Barov et al. (2022) as a species “taking large areas, impact on pollinators unknown”. It can form dense stands in natural grasslands and outcompete native plants. It is also known to change soil chemistry, natural succession and increase the fire risk (e.g. Domènech et al. 2006). No evidence of significant negative impacts on pollinators could be found in Europe or elsewhere (CABI 2009b).</td></tr>
<tr><td>Elaeagnus angustifolia</td><td>Silver berry</td><td>Greater Middle East</td><td>17,958</td><td>2</td><td>Outcompeting native plants; Habitat modification</td><td>1</td><td>Barov et al. (2022)</td></tr>
<tr><td colspan="8">Introduced to North America and Europe. Invasive in riparian, floodplain and wetland habitats, also in salt marshes and dunes. It displaces native vegetation and modifies the habitat by nitrogen fixation, interfering with natural succession, and reducing bird and insect diversity (Katz & Shafroth 2003, Sudnik-Wójcikowska et al. 2009, CABI 2018). The flowers produce nectar and pollen, which attract and are pollinated by honeybees and bumblebees. It is considered sometimes as a valuable honey plant. Pendleton et al. (2011) found lower numbers of pollinating insects on the plant compared to native shrubs, but also noted a lack of pollination studies for this species.</td></tr>
<tr><td>Erigeron bonariensis (= Conyza bonariensis)</td><td>Hairy fleabane</td><td>Central and South America</td><td>17,246</td><td>2</td><td>Outcompeting native plants</td><td>1</td><td>Barov et al. (2022)</td></tr>
<tr><td colspan="8">Introduced to North America, Africa, Asia, Australia, New Zealand and Europe. It is an important – and difficult to control – weed in orchards, vineyards, crops, and pastures, has allelopathic effects, competes for water and nutrients with other plants and is host to different plant pests. It is generally considered less problematic and casual in natural habitats (Wu 2007, Bajwa et al. 2016, CABI 2021). No evidence of significant negative impacts on pollinators could be found in Europe or elsewhere (Info Flora 2022).</td></tr>
</table>

Species	Common name	Native range	# European records (GBIF)	D	Impact	I	Source
Erythranthe moschata (= Mimulus moschatus)	Musk monkey-flower	North America	485	1	Outcompeting native plants; Toxic (alkaloids as secondary compounds)	2	Truscott et al. (2008b), Baude et al. (2011)*
Introduced to South America, South Africa, Tasmania and Europe. Colonises disturbed areas on riverbanks (e.g. Truscott et al. 2008a, Koopman et al. 2012). According to Hejda & Pyšek (2008) and Hejda et al. (2009), it has a low impact on native species richness. Truscott et al. (2008b) found a negative association between *Mimulus* cover and native and non-native plant species richness, including native riparian plant species displacement at the local scale. Baude et al. (2011) found, in a pot experiment, higher soil nitrogen acquisition and competition with the native *Lamium amplexicaule*, having a negative impact on nectar amount, quality and floral display on the native species, potentially reducing attractiveness to shared pollinators.							
Fallopia japonica (= Reynoutria japonica)	Japanese knotweed	East Asia	170,351	3	Outcompeting native plants; Habitat modification	1	Davis et al. (2018)
Fallopia japonica, and other knotweed species, are well-known widely distributed invasive species in North America and Europe (e.g. CABI 2013b). *Fallopia japonica* is considered one of the “100 worst”, building monodominant stands, outgrowing, overshadowing and replacing native plant species and changing plant and soil communities via allelopathic effects. Negative impacts on abundance, species richness and biomass of invertebrates are documented (Gerber et al. 2008), with knock-on effects on some vertebrates (e.g. Maerz et al. 2005). Davis et al. (2018), however, found higher flowering plant species richness and abundance and higher abundances of bumblebees, higher diversity of hoverflies and higher overall insect diversity at invaded than uninvaded sites, probably due to late flowering and the paucity of other flowering species at this time of the year. It is also considered sometimes as a valuable honey plant.							
Galinsoga parviflora	Quickweed	South America	43,332	2	Outcompeting native plants	1	Barov et al. (2022)
Barov et al. (2022) mention “*Bidens parviflora*”, a native of eastern Asia that is not introduced elsewhere, but presumably mean *Galinsoga parviflora*, which is introduced to all continents. It is an annual herb, competitive in disturbed habitats and an economically relevant weed in crops, gardens and greenhouses. No evidence for significant negative impacts on pollinators could be found in Europe or elsewhere (CABI 2014b).							
Heracleum mantegazzianum[i]	**Giant hogweed**	Caucasus	148,325	3	Outcompeting native plants; Habitat modification	3	Nielsen et al. (2008), Zumkier (2012)*, Davis et al. (2018), Rabitsch et al. (2021), Bogusch et al. (2023)
Impatiens glandulifera[i]	**Himalayan balsam**	Himalayas	417,566	3	Outcompeting native plants	3	Prowse & Goodridge (2000)*, Chittka & Schürkens (2001)*, Lopezaraiza-Mikel et al. (2007), Nielsen et al. (2008)*, Nienhuis et al. (2009)*, Vilà et al. (2009), Bartomeus et al. (2010), Cawoy et al. (2012)*, Thijs et al. (2012), Emer et al. (2015)*, Davis et al. (2018), Rabitsch et al. (2021), Barov et al. (2022)

<table>
<tr><th>Species</th><th>Common name</th><th>Native range</th><th># European records (GBIF)</th><th>D</th><th>Impact</th><th>I</th><th>Source</th></tr>
<tr><td>Lupinus nootkatensis</td><td>Nootka lupine</td><td>North America</td><td>2,465</td><td>1</td><td>Outcompeting native plants; Habitat modification; Toxic</td><td>2</td><td>Willow et al. (2017)*</td></tr>
<tr><td colspan="8">Introduced to and widespread in Iceland, established also in Scandinavia. Although floral displays are larger than those of native plants in Iceland, it was visited less by insects, including the only native bumblebee, which is under threat if native vegetation is replaced by the alien plants (Willow et al. 2017). It has been shown that secondary compounds (lupanine) are toxic to bumblebees (Arnold et al. 2014). Nitrogen-fixing and driving vegetation dynamics in nutrient-limited habitats, e.g. coastal dunes (Hanslin & Kollmann 2016).</td></tr>
<tr><td>Lupinus polyphyllus</td><td>Large-leaved lupine</td><td>North America</td><td>202,529</td><td>3</td><td>Habitat modification</td><td>3</td><td>Valtonen et al. (2006)*, Jakobsson & Padrón (2014)*, Jakobsson et al. (2015)*, IUCN (2020)</td></tr>
<tr><td>Mahonia aquifolium (= Berberis aquifolium)</td><td>Oregon grape</td><td>North America</td><td>52,539</td><td>2</td><td>Outcompeting native plants; Habitat modification</td><td>1</td><td>Barov et al. (2022)</td></tr>
<tr><td colspan="8">Introduced to Europe as an ornamental and honey plant in the early 19th century. It outcompetes and displaces native vegetation and changes natural succession (e.g. Auge & Brandl 1997, Ross & Auge 2008, CABI 2010). It is pollinated by insects and is a praised source of pollen and nectar for bumblebees in winter. Competition between honeybees and wild bees in urban habitats cannot be ruled out (e.g. Renner et al. 2021), but no evidence for significant negative impacts on pollinators could be found in Europe or elsewhere. It is considered sometimes as a valuable honey plant.</td></tr>
<tr><td>Miscanthus spp.</td><td>Silvergrass</td><td>South Africa, Trop. and temp. Asia</td><td>2,881</td><td>1</td><td>Habitat modification</td><td>1</td><td>Barov et al. (2022)</td></tr>
<tr><td colspan="8">Mentioned by Barov et al. (2022) as a species “taking large areas, impact on pollinators unknown”. It can form dense stands in natural grasslands and outcompete native plants (e.g. Schnitzler & Essl 2015), but no evidence for significant negative impacts on pollinators could be found in Europe or elsewhere.</td></tr>
<tr><td>Nicotiana glauca</td><td>Tree tobacco</td><td>South America</td><td>6,959</td><td>1</td><td>Habitat modification; Toxic (alkaloids as secondary compounds)</td><td>2</td><td>Ollerton et al. (2012)*, Kasiotis et al. (2020)*, Barov et al. (2022)</td></tr>
<tr><td colspan="8">Mentioned by Barov et al. (2022) as a species with “dense stands, nectar, not attracting pollinators, climate change favoured”. Ollerton et al. (2012) and Issaly et al. (2020) found that in regions without bird pollinators (such as in Greece and Mallorca), seeds are set by selfing and the nectar is only rarely exploited by bees or butterflies. It can form dense, monospecific stands in semi-arid disturbed habitats, and displace native species (CABI 2013a). Leaf litter of this plant inhibits germination of native plants (Florentine & Westbrooke 2005) and it shifts the soil microbial community composition (Rodríguez-Caballero et al. 2020). The plant is toxic (containing anabasine) to humans and livestock, and caused up to 25% mortality in feeding experiments in honeybees (Kasiotis et al. 2020). The significance on pollinators in the environment in Europe remains unclear.</td></tr>
</table>

Species	Common name	Native range	# European records (GBIF)	D	Impact	I	Source
Nicotiana tabacum	Tobacco	South America	2,621	1	Toxic (alkaloids as secondary compounds)	2	Detzel & Wink (1993)*, Stevenson et al. (2017)*
Cultivated around the world and escaped as an agricultural weed in many regions, including Europe. While the toxicity of the plant is well known, including possible effects on insect behaviour (e.g. Baracchi et al. 2017), there is a lack of studies on its impact on biodiversity and on pollinators in Europe (CABI 2014c).							
Oenothera speciosa and *O. biennis*	Pink evening primrose, Common evening primrose	North America	973 (*O. speciosa*) 40,777 (*O. biennis*)	1 2	"Mothcatching"	1	Zlatkov et al. (2018), Barov et al. (2022)
Oenothera speciosa is a garden escape that has established in Asia, Australia, and Europe (AL, BE, BG, ES, FR, GR, HU, IT, UK). It has large, showy flowers that are frequently visited by different insects, especially by moths. Zlatkov et al. (2018) have shown that hummingbird hawk-moths can get stuck in the flowers and subsequently die, but there is no evidence of a significant negative effect on pollinators at the population or community level in Europe. *Oenothera biennis* has a wide distribution in Europe. *Oenothera species* are insect-pollinated (e.g. Antoń et al. 2017), and they prefer dry and disturbed habitats. They rarely grow in dense stands and rarely outcompete native plants (Mihulka et al. 2006).							
***Opuntia* spp. (e.g. *O. ficus-indica*, *O. stricta*)**	**Prickly pear**	North and South America	75,429	2	Outcompeting native plants; Habitat modification	3	Bartomeus et al. (2008a), Padrón et al. (2009), Vilà et al. (2009)
Oxalis pes-caprae	**Sourgrass**	South Africa	32,243	2	Outcompeting native plants	3	Jakobsson et al. (2009), Albrecht et al. (2016)
Prosopis juliflora[i]	Mesquite	Central and South America	1	1	Outcompeting native plants; Habitat modification	1	Barov et al. (2022)
Introduced as fuel and fodder in Asia, Africa, Australia and present in Europe in the wild in Spain and on Gran Canaria (only one GBIF record from Gran Canaria). It is nitrogen-fixing, modifies water availability and natural succession and can outcompete and replace native plants (CABI 2017, EPPO 2018). It is pollinated by insects, especially by bees. There is no evidence of a significant negative effect on pollinators at the population or community level in Europe.							
Rhododendron ponticum	**Pontic rhodo-dendron**	Iberia, Caucasus	62,581	2	Outcompeting native plants; Toxic	3	Stout (2007), Vilà et al. (2009), Dietzsch et al. (2011), Stout & Casey (2014)*, Tiedeken et al. (2014*, 2016), Tiedeken & Stout (2015)*, IUCN (2020), Barov et al. (2022)

Species	Common name	Native range	# European records (GBIF)	D	Impact	I	Source
Robinia pseudoacacia	**Black locust**	North America	236,853	3	Habitat modification	2	Buchholz et al. (2015)*, Reif et al. (2016)*, Buchholz & Kowarik (2019), Barov et al. (2022)
Senecio inaequidens	South African ragwort	South Africa	345,003	3	?	1	Vanparys et al. (2008, 2011*)
Introduced to Europe with wool imports in the late 19th century, it is widespread and still spreading further. It is found in ruderal and disturbed habitats. It is toxic to mammals (CABI 2014d). Vanparys et al. (2008) showed that the visitation rate by pollinators and the seed set of *S. inaequidens* were higher compared to that of the native relative *Jacobaea vulgaris (Senecio jacobaea).* Vanparys et al. (2011) found, however, that the presence of *S. inaequidens* did not alter the pollinator visits and seed set of *J. vulgaris*, concluding that other traits need to be investigated to explain the different visitation rates and reproductive success between the two species.							
Solanum elaeagnifolium	Silverleaf nightshade	North America	2,337	1	Outcompeting native plants; Habitat modification	2	Tscheulin et al. (2009), Tscheulin & Petanidou (2013)
Unintentionally introduced as contaminant of fodder crops and packing material and unintentionally spread by livestock, contaminated soil, agricultural vehicles and machinery. Isolated occurrences in Europe (Roberts & Florentine 2022, Tataridas et al. 2023). Tscheulin et al. (2009) examined the effect of *S. elaeagnifolium* invasion on flower visitation patterns and seed set of the co-flowering native *Glaucium flavum* (Papaveraceae) and found that *G. flavum* flowers in uninvaded sites received significantly more total visits from pollinators. Tscheulin & Petanidou (2013) found that, in the presence of potted *S. elaeagnifolium* plants, pollen limitation was significantly enhanced, although the overall visitation rates were not reduced, due to a reduction in honeybee visitation in the presence of the invasive resulting in reduced pollination.							
***Solidago* spp. (*S. canadensis* and *S. gigantea*)**	**Canadian and Giant goldenrod**	North America	148,898 (*S. canadensis*) 157,364 (*S. gigantea*)	3	Habitat modification	3	de Groot et al. (2007)*, Moron et al. (2009), Fenesi et al. (2015), IUCN (2020), Barov et al. (2022)
Spiranthes cernua x odorata		?	?	1	?	2	Barov et al. (2022)
Mentioned by Barov et al. (2022) as a species “growing on meagre grasslands and sand, abundant nectar, evidence of its impact to be published soon, attracts native plant pollinators; early invasions stage in the Netherlands”. The taxonomy of the *Spiranthes cernua* complex, however, is challenging and Pace & Cameron (2017) found no evidence of hybridisation between the North American *S. cernua* and (the relatively distantly related) *S. odorata.*							

3.2.1. Relevant species and species ranking

Table 3 presents the matrix of IAPS based on the scoring of their geographic distribution and impact on pollination in Europe. Species highlighted in red and all remaining species in the impact category 3 are described in more detail in Section 5, as being considered those currently presenting more risks to pollinators and pollination in the EU.

Table 3: Matrix of IAPS considering scoring of distribution and impact on pollination in Europe.

		Impact		
		1	2	3
Distribution	1	*Amorpha fruticosa* *Baccharis halimifolia* *Catalpa speciosa* *Miscanthus* spp. *Oenothera speciosa* *Prosopis juliflora*	*Erythranthe moschata* *Lupinus nootkatensis* *Nicotiana glauca* *Nicotiana tabacum* *Solanum elaeagnifolium*	*Asclepias syriaca*
	2	*Araujia sericifera* *Arundo donax* *Cortaderia selloana* *Elaeagnus angustifolia* *Erigeron bonariensis* *Galinsoga parviflora* *Mahonia aquifolium* *Oenothera biennis*		*Carpobrotus* spp. *Opuntia* spp. *Oxalis pes-caprae* *Rhododendron ponticum*
	3	*Fallopia japonica* *Senecio inaequidens*	*Robinia pseudoacacia*	*Buddleja davidii* *Heracleum mantegazzianum* *Impatiens glandulifera* *Lupinus polyphyllus* *Solidago* spp.

4. Non-chemical management methods for terrestrial invasive alien plant species

4.1. Available methods

The following sections present the most effective and used non-chemical management methods against IAPS. The information presented includes a general description of the methods, good implementation practices, notes on their effectiveness, potential costs and resources needed, side effects, and indication of feasibility or shortcomings of implementation. Information on which of the 11 most harmful (groups of) species the method can be used for is also provided. General considerations that should be taken into account when using physical/mechanical methods are highlighted, as well as some notes on the importance of using chemical methods in specific circumstances. Finally, the description of all non-chemical methods is summarised in Table 4 below.

4.1.1. Covering soil with plastic sheets

General description

This method involves placing a dense plastic cover (usually polyethylene black in colour) over the infested area and has been used, for example, in Danish management actions on giant hogweed (Suadicani et al. 2017). The cover has to be made from dense plastic, to eliminate light. It is important to fix the cover firmly on the ground to prevent any movement due to wind. For species that depend on generative reproduction, the cover must be placed over the species in early spring before germination. An analogue approach is to cover high stumps after cutting with plastic sacks (e.g. for *Robinia pseudoacacia*) to reduce resprouting. For species where the management is targeted against germinated seedlings, the plastic cover can be removed in August of the same year. It is important to revegetate the managed site, ideally with native species. Revegetation can be done by direct seed sowing, or by spreading local hay or mulch.

It is not recommended to leave the plastic at a site for a period longer than one vegetation season, as the additives used in plastic could leach into the environment and, furthermore, to stop the possible release of microplastics.

Covering Ice plant (*Carpobrotus edulis*) by mulching sheets. Conserving threatened habitats and species in Berlengas SPA through sustainable management, LIFE Berlengas (LIFE13/NAT/PT/000458)

The results after Ice plant (*Carpobrotus edulis*) has been covered with mulching sheets. Conserving threatened habitats and species in Berlengas SPA through sustainable management, LIFE Berlengas (LIFE13/NAT/PT/000458)

Good implementation practices

The plastic cover must be fixed to the ground properly to eliminate any movement by wind and later damage of the cover. The site has to be checked regularly to fix any possible damages (rift) of the foil that may decrease its efficiency.

Effectiveness

The method is effective for small areas and species without underground storage organs. For highly vigorous species with rhizomes (rhizomatous species), this method is not efficient, as e.g. *Fallopia* sp. can survive under the cover for more than three years with no effect. This method is also not suitable for woody species, as the stumps may destroy the cover.

Costs, effort and resources required

Materials needed include the plastic cover and tools to fix it to the ground. In the management of giant hogweed in Denmark, 400 m² were covered per hour and an estimated cost of 25.000 SEK (approximately 2.200 EUR) was needed per hectar. As the cover is placed early in the season, the risk associated with potential contact with the sap and skin burns (phytophotodermatitis) is low. The advantage is that the method is suitable even for organic farmers. This method is feasibly applied for areas up to 100-200 m² (Rajmis et al. 2016), and there are no additional costs identified.

Potential side effects

The use of plastic can lead to environmental issues like plastic waste accumulation and soil degradation. Over time, the plastic may break down into microplastics, which can harm the ecosystem and potentially contaminate water sources. When removing the plastic, the overall aboveground vegetation has been usually targeted. Therefore, it is necessary to properly manage the site after the cover has been removed to aid revegetation. It is also important to prevent reinvasion from neighbouring areas and from the seed bank.

Feasibility and potential shortcomings

This method is applicable to small areas in grassy vegetation or in sites where woody species have been cut to low stumps. It is not suitable for heterogeneous sites or sites which have other invasive species in neighbouring areas, as the method creates open habitat ready for reinvasion.

Most harmful IAPS that can be managed

Carpobrotus spp., *Heracleum mantegazzianum*, *Impatiens glandulifera*, and *Oxalis pes-caprae*

4.1.2. Pulling

General description

The recommended method is to pull out the whole plant with its roots, rhizomes and bulbs. Mature or regenerating plants should be pulled out, at the latest, in the early stages of the flowering period to prevent seed formation. Mature seeds shoot out into the surroundings during handling, and immature seeds are able to mature even after the plants have been pulled out and thus complicate biomass handling. The treated areas should be checked after the intervention and during the season (after about 2-3 weeks) and the remaining individuals should be pulled out. Subsequent repeated control is necessary for several years. Zero tolerance when removing all individuals and proper treatment (removing all the organs capable of resprouting) are important for the action to be effective, along with preventing seeds from spreading to the site. Handling of plant material must be done to prevent unintentional spread by plant waste or seeding at the site. Biomass can only be left at the site if the intervention has occurred before flowering and the plants have not yet started fruiting. At the same time, re-rooting of the plants must be prevented, e.g. by separating the roots from the stem and breaking the uprooted plant so as to minimise the ability of the plant to regenerate and form adventitious roots from the nodes.

Ice plant (*Carpobrotus edulis*) removal. Invasive species control through public participation, LIFE Biodiscoveries (LIFE13/BIO/PT/000386) © 2020 Lúcia Santos. All rights reserved. Licenced to the European Union under conditions.

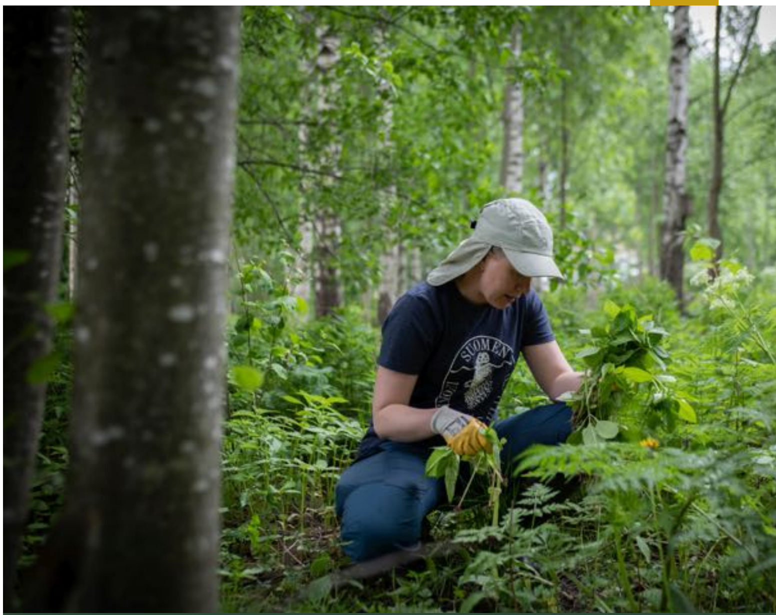
Pulling can be used for both annual species (e.g. *Impatiens* spp.) and perennial species. For annual species this is a suitable strategy due to the usually shallow and simple root system and high efficiency. Pulling of perennial species (herbaceous or woody) is only effective on young seedlings that do not have a branched root or rhizome system (e.g. *Buddleja davidii, Prunus serotina*). For perennial species, it is usually effective on young plants or plants in a sandy substrate. In perennial or repeatedly mown clumps, it is usually not possible to remove all roots, but pulling can significantly reduce the species vitality and density and thus be suitable for density reduction, even if it usually does not lead to complete eradication. Pulling of large shrubs and trees is discussed in section *4.1.6. Digging up the underground organs.*

Good implementation practices

Species capable of regenerating must not be, after removal, disposed to sites that allow their regeneration (rooting). For some species (e.g. *Impatiens glandulifera*), the biomass may be left in place, but only if the disturbance occurs before flowering, so the plants do not have pollinated flowers from which germinating seeds can develop. Pulled plants can be placed in safe locations, e.g. in sunny areas away from watercourses or other sources of moisture to prevent the plants from regenerating, or placed on top of surrounding taller vegetation. In the case of large quantities of managed plants, pulled plants can be collected on a plastic sheet/foil and then mechanically damaged. The plastic sheet should be removed at the end of the growing season to prevent its damage by frost.

Effectiveness

In general, pulling is a very effective method. It can be applied both to small sites and large sites with low density of targeted species. Large sites with high density are demanding on labour effort. The method creates new disturbed sites and therefore inappropriate biomass management practices and possible propagule transport must be avoided. For some species with a high capacity to regenerate from below-ground biomass (e.g. *Fallopia* spp., *Asclepias* spp.), this method is inappropriate because it is too risky in terms of further spread. If it is resorted to, all belowground biomass must be separated from the soil, dried and burned. The method is also effective for perennial species intolerant to disturbance of aboveground biomass and which do not react to the disturbance by resprouting (e.g. appropriate for *Solidago* spp., but not for *Asclepias* spp.). The method is very efficient for low abundance or rare occurrences of IAPS at the borders of their distribution. It is cheap and rapid. It can also be used for species like *Senecio inaquidens* or *Ambrosia artemisiifolia*. Small patches of e.g. *Lupinus polyphyllus* can be, under repeated management, eradicated by pulling through weakening, even if the whole roots are not excavated.



Volunteer work coordinator removing Himalayan balsam (*Impatiens glandulifera*). Awareness building, surveying and controlling invasive alien species (IAS) in Finland, Finvasive LIFE (LIFE17/NAT/FI/000528) © 2022 Titta Vikstedt. All rights reserved. Licenced to the European Union under conditions.

Costs, effort and resources required

This method is not demanding in terms of equipment, but it is costly in terms of human labour. The use of volunteers is possible, but the quality of work must be checked, as usually when pulling the perennial plants volunteers remove only the aboveground biomass and roots remain undisturbed. Proper practice and control is therefore needed, as well as training of volunteers.

Potential side effects

When done properly there are no side effects, except for possible soil disturbances in the invaded area.

Feasibility and potential shortcomings

The method is easy to apply.

Most harmful IAPS that can be managed

Seedlings of woody species like *Buddleja davidii, Rhododendron ponticum* and *Robinia pseudoacacia;* herbaceous species*: Asclepias syriaca* (only seedlings), *Carpobrotus* spp., *Impatiens glandulifera, Lupinus polyphyllus, Opuntia* spp., *Oxalis pes-caprae,* and *Solidago* spp.

4.1.3. Heat treatment

General description

This method is based on exposing the seedbank and live organs of the plants to heat. Heat can be produced by different ways including fire, hot foam/ water or microwaves. The methods, if applied at a small scale, seem to be rather environmentally friendly, as they are not associated with strong side effects, but the energy needed to sufficiently heat the soil or seeds is extreme. Therefore, this limits the use of these heat treatments in the field. The majority of studies arise from laboratory experiments. Generally for seeds, temperature equal to or higher than 90°C does not completely inhibit their germination and allows 1% or 2% germination. The effectiveness increases with an increase in the duration of the treatment, as after a longer exposure (i.e. 30 minutes) to a hot water bath at 50°C and higher, germination usually decreases significantly (e.g. Oliver et al. 2020). The effect of heat treatment is comparable to composting (Strgulc Krajšek et al. 2020).

1. Foam

Hot foam is based on hot water treatment, but is modified by the addition of biodegradable foaming agents. The foam insulates the weeds from the surrounding air and increases the energy transfer to the plants, thus lowering the dose of hot water required and increasing efficiency.

2. Microwaves

Microwave radiation causes dielectric heating of moist materials and offers a means of rapidly reaching the temperature needed for loss of seed viability and inhibition of germination (60 to 90°C range). Microwaves are used to control invasive and pest species for commercial, agricultural, or ecological purposes, particularly in the interest of developing non-chemical techniques.

3. Direct heating - fire

Directed heating is based on applying intense heat by propane torch or a similar tool. It is different to prescribed burning because it is targeted at the species being controlled. Direct heating can often destroy woody invasive plant seedlings. However, single instances of spot burning are generally not effective for controlling mature woody invasive alien species due to their ability to regrow from the roots.

Good implementation practices

For direct heating, heat directed at the roots and leaves of seedlings usually causes sufficient damage to prevent regrowth. When applied to mature plants, stems are flame-girdled, killing the aboveground growth. Frequent repeated spot burning of regrowth may eventually lead to mortality, though the frequency and duration required are often unknown and likely vary between species and ages of target plants.

Effectiveness

This method is only effective for species with shallow root system or young seedlings due to limits of heat transmission to deep soil layers. The method destroys the upper organs (leaves). The foam heat treatment is not effective for seeds, but it can be used for seedlings that are sensitive to changes in temperature. Direct fire or microwaves can affect seeds.

Costs, effort and resources required

The equipment needed for the microwave treatment is costly, as well as the operating costs. In addition, the available equipment is primarily designed for use in laboratory work. The methods depend on supply of energy. A propane torch for direct burning is relatively cheap.

Potential side effects

The method, in general, has no direct environmental drawbacks, especially as there is no residue to contaminate the surroundings, unlike herbicides. However, results of experiments have revealed the low applicability of this method in the field (e.g. microwave treatments), which decreases overall effectiveness and which is costly. Use of the open fire can be limited by local environmental restrictions due to risk of the fire spreading. This method may also induce germination of some species that require heat treatment for the seeds to germinate and it may disturb the aboveground vegetation and thus allow reinvasion.

Feasibility and potential shortcomings

The method needs a high amount of energy to sufficiently heat the soil or seeds, therefore it is used only in easily accessible sites (e.g. agriculture land).

Most harmful IAPS that can be managed

Carpobrotus spp., *Heracleum mantegazzianum, Impatiens glandulifera, Lupinus polyphyllus, Oxalis pes-caprae,* and *Solidago* spp.

4.1.4. Girdling

General description

Girdling (ring-barking) is a method applicable to woody invasive alien plants and is based on removing a strip of bark around the entire (full girdling) or partial (ca. 80%) circumference of the plant's trunk or stems. The method is therefore practicable only on relatively large single-stemmed plants. Girdling a tree involves removing the protective outer layer of bark and the vascular tissue directly under the outer layer in a ring around the entire trunk. Girdling disrupts the ability of a woody plant to move water and sugars between the roots and the shoots/stems and eventually kills the plant above the girdle, though regrowth may occur below the girdle.

Good implementation practices

Partial ringing consists of removing a strip of bark (e.g. about 20 cm wide in mature tree) at a height of 1 to 1.5 m, down to a depth of about 2 cm. The depth must be sufficient to remove the vascular cambium, i.e. the thin layer of living tissue in which nutrients move between leaves, roots and growth cells. To reduce resprouting, it is important to not complete the ring and to leave a few centimetres wide strip of border (about 15-20%), which allows partial nutrient flow. The tree is thus gradually thinned out and does not become as massively thin as if the ring were made around the entire circumference of the trunk. If the tree is not regenerating widely, partial ringing can be closed in the second year and cutting the tree in the third year.

Resprouting from the roots and below the ringed area on *Ailanthus altissima* tree after ring-barking method was not correctly applied © HermannFalkner/sokol, (CC BY-NC 2.0) via Flickr

Effectiveness

The method reduces the regeneration of resprouting woody species. This method is partly effective and, although less effective and more time consuming compared to herbicide application, it can be used in areas where herbicides are not allowed. Girdling alone is often ineffective for long-term control of woody invasive species due to their ability to regrow from the root system. Therefore, it is important to use partial girdling, which reduces tree regeneration from roots, compared to full girdling. Healthy, mature trees can take several years to die if girdled. Although trees usually regenerate less after partial girdling than after felling, girdling usually results in intensive rejuvenation requiring further intervention.

Costs, effort and resources required

The girdling method is demanding to properly execute, in terms of time and tools needed, and is therefore not widely used at present. It is also less practical, especially when direct herbicide application to drilled holes is an available option (see Section *4.4. Notes on using chemical management methods*).

Potential side effects

Human labour and training is needed to properly girdle the trees. If done incorrectly, rapid growth of suckers can be expected, which therefore worsens the invasion.

Feasibility and potential shortcomings

The method is feasible, but good training on the application of the method is needed.

Most harmful IAPS that can be managed

Rhododendron ponticum and *Robinia pseudoacacia* (although the method is not effective enough)

Ring-barking, Invasive species control through public participation, LIFE Biodiscoveries (LIFE13/BIO/PT/000386)

4.1.5. Cutting

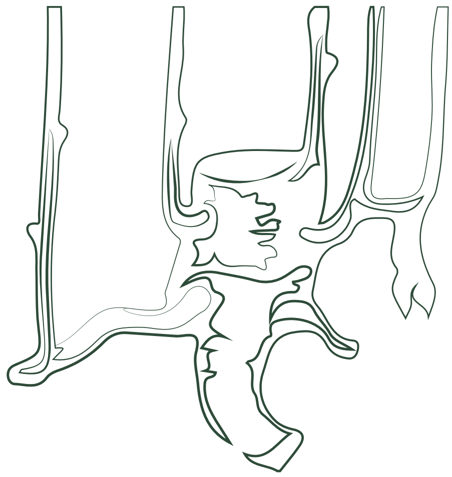
General description

Easy resprouting of *Ailanthus altissima* after cutting © Jan Pergl

Cutting is a method suitable for non-resprouting woody plants. For all trees with vigorous vegetative reproduction, cutting alone, which produces a dense growth of shoots, must be replaced by mechanical intervention combined with targeted application of herbicides or, if not possible to use herbicides, partial girdling.

1. Sprout control as a subset of cutting

This consists of the mechanical removal of sprouts of invasive woody plants by machete, brush scythe, brush saw or chainsaw. Timing is crucial to prevent resprouts from becoming woody. It can also be used before grazing to create a homogeneous population of fresh resprouts. This method is used in habitats like under electricity lines and pylons, along railways and roads due to being easier to apply (low demand on personnel training). However, to maintain the site in the same condition, it has to be repeated regularly, almost every year. The method is not recommended for final and sustainable management of IAPS.



Good implementation practices

Cutting on tall stumps (ca 1-1.5 m) is used to reduce the formation of coppice where, for safety reasons, ringing or drilling with herbicide injection cannot be used, leaving the tree to die. Cutting (large trees or sprouts) can be used for homogenisation of the stand before the following and final management actions (e.g. herbicide spraying).

Effectiveness

This method is effective for non-resprouting woody species like *Pinus nigra* and *P. strobus*. It is not recommended for species (trees and shrubs) with root or stump regeneration (e.g. *Robinia pseudoacacia*, *Rhododendron ponticum* and *Acer negundo*). For these species, it is better to use girdling, if use of herbicides is not allowed.

Costs, effort and resources required

This method is cheap and straightforward to apply, with no special equipment required.

Potential side effects

If applied to resprouting species, then the species develops dense and large stands of resprouts at the site, which are almost impossible to eradicate. Taking the example of *Ailanthus altissima*, if the adult tree is just cut down, it resprouts quickly from the roots and stump, and creates a dense stand which is costly to eradicate. Partial girdling reduces the number of resprouts, but does not kill the individual.

Feasibility and potential shortcomings

This is a relatively easy method to apply. If applied to resprouting species, then the species develops dense and large stands of resprouts at the site, which are almost impossible to eradicate.

Most harmful IAPS that can be managed

This method is not recommended for any of the listed species as a method of management that leads to final eradication. However, it can be used for controlling the spread of *Buddleja davidii* by reducing its seed set. For *Rhododendron ponticum*, it can be used to create access to sites where stands are overgrown by the branches, for subsequent herbicide application, or for control of spread.

Removal of an *Acacia* tree by cutting. Conserving threatened habitats and species in Berlengas SPA through sustainable management, LIFE Berlengas (LIFE13/NAT/PT/000458) © 2021 SPEA. All rights reserved. Licenced to the European Union under conditions.

4.1.6. Digging up the underground organs

General description

Digging up the roots covers two approaches: digging the roots of rhizomatous species and digging of relatively compact roots, mostly shrubs and trees. Digging rhizomatous roots is very laborious and usually does not lead to the complete eradication of the species on the site. Digging up the underground organs can be replaced by ploughing in some occasions, such as at infested agricultural land and only for species which have limited resprouting capacity. Ploughing can be used for mechanical preparation of infested sites by splitting the underground organs for following control by other techniques. Ploughing is only marginally effective (only effective on seedlings), but can stimulate resprouting even in herbaceous species (e.g. *Asclepias syriaca*). Ploughing also has the disadvantage of creating large disturbances suitable for seedlings from unmanaged invaded areas to establish. Moreover, both methods carry the risk of further spreading the species through plant material left on machinery or in contaminated soil.

On the other hand, removal with an excavator can be used for non-regenerating species, but also for trees and shrubs that form root suckers (e.g. *Ailanthus altissima*, *Robinia pseudoacacia*, *Acer negundo*, *Amorpha fruticosa* and *Budleja davidii*). In this case, the underground roots have to be removed deeply and widely. If digging is chosen, all belowground biomass must be separated from the soil, dried and burned.

Good implementation practices

Annual plants and some shallow-rooted perennial plants can be removed with their roots by scraping the soil, which removes the top layer of soil and much of the seed bank. Vines, deep-rooted perennial forbs and grasses, shrubs, and trees more often require deeper and more targeted excavation. This technique can be used for eradication of IAPS, but it must be followed by efforts to revegetate a site and ensure its seed bank is fully depleted. If eradication is the goal, this measure should be combined with other measures aimed at detecting and treating (mainly removing) emerging seedlings, for instance, from missed propagules or ineffective treatments, to ensure its long-term effectiveness.

Effectiveness

The method is very effective for species with a compact undergrown root/rhizome system. An added advantage is the absence of stumps that would hinder follow-up care. For species with fragile and long roots/rhizomes it is not recommended (e.g. *Fallopia* sp., *Asclepias syriaca*). Even for regenerating tree and shrub species (e.g. *Robinia* spp. and *Ailanthus altissima*) it can be used if done properly and possible resprouts managed in following years.

Costs, effort and resources required

It requires a large amount of human labour for large areas. For rapid eradication of a few individuals, this method is possible. The cost of mechanisation work is high. It is necessary to treat and deposit the contaminated soil in an appropriate place.

Potential side effects

A disadvantage of this method is the amount of contaminated soil that needs to be dealt with. This method is most effective at controlling small invasive species populations, as the removal of large invasive species populations would require extensive effort, create large-scale disturbances and pose issues associated with the disposal of large above- and below-ground biomass.

Feasibility and potential shortcomings

For some species with a high capacity to regenerate from below-ground biomass, this management method is inappropriate because it is too risky in terms of further spread. For species without seedbanks or without rhizomes, the soil can be freely used. For species with seeds below the tree or with a rich rhizome/root system, the soil has to be moved, deposited and used with caution. Digging is sometimes recommended for *Fallopia* spp. or similar species, but the amount of contaminated soils, deep and wide rhizome system and risk of unintentional spread due to soil transport is very high.

Most harmful IAPS that can be managed

Seedlings of *Asclepias syriaca, Buddleja davidii, Carpobrotus* spp. *Heracleum mantegazzianum, Lupinus polyphyllus, Opuntia* spp., *Oxalis pes-caprae, Rhododendron ponticum, Robinia pseudoacacia,* and *Solidago* spp.

Removal works. DUNe restoration by tackling Invasive Alien Species, LIFE DUNIAS (LIFE20/NAT/BE/001442) © 2020 Reinhardt Strubbe. All rights reserved. Licenced to the European Union under conditions.

4.1.7. Grazing

General description

Grazing in fenced pastures is suitable for large areas and areas under organic farming. For grazing, various kinds of animals can be used. When grazing is applied, the carrying capacity of the site has to be taken into account to set the amount of animals needed. Grazing is less effective than mowing because it is less regular and the grazing itself depends on the behaviour of the animals. The whole pasture needs to be controlled for non-grazing individuals.

Good implementation practices

Grazing needs to be carried out in such a way as to ensure the appropriate length and timing of grazing, the size of the herd and the appropriate type of livestock. It is important to consider grazing characteristics and livestock weight when planning management using this method. The grazing intensity must be chosen with regard to the carrying capacity of the land to avoid damaging the vegetation cover and reducing soil erosion. Grazing must be started before the plants or shoots become woody. If needed, resprouting woody species can be cut to become fresh (see section *4.1.5. Cutting*). It must be timed so as to avoid the formation of seeds, which animals may spread in their droppings or on their fur. Areas, and edges of areas, not grazed must be subsequently and timely managed (i.e. before seed production). This practice is recommended for minimising reinvasion of land intended for organic farming, after it has been treated with other methods (e.g. herbicides). Grazing can also be introduced after spraying herbicides to control emerging seedlings, but not before the expiry of the protective period of the product used.

Effectiveness

This method is effective for large invaded areas and for reducing the seed set. Grazing can be used to reduce the density of IAPS on large tracts of land, but rarely leads to complete eradication of species. In addition, some of the IAPS here described are either unpalatable or respond quickly to disturbance through vegetative growth and/or germination, so this method would not be effective for those (see examples below).

Costs, effort and resources required

Grazing must be applied repeatedly over several years. A one-off intervention is not recommended due to the minimal effect and large disturbances that will lead, in the majority of cases, to an increase in invasive species density. The costs for grazing depend on the local availability of the animals, need for transport, etc. The costs will differ for areas regularly managed by grazing or sites that are grazed irregularly. Irregular pastures need extra fencing, transport of animals to the site and additional equipment like water or shelter infrastructure.

Grazing goats © gpparker, (CC BY-NC 2.0) via Flickr

Potential side effects

Using animals in management has to be considered in the context of the invaded habitat, as the occurrence of animals is associated with nitrogen increase and change of disturbance regime. There is a risk of seeds or other propagules being spread by animals on coats, fur and, in addition, of seeds being incorporated into the soil by their hooves.

Feasibility and potential shortcomings

It is important to consider the diversity of species available to the grazing animals, ensuring that they have access to species other than the invasive ones. Some invasive plants can be toxic if ingested (e.g. *Asclepias syriaca* and *Prunus serotina*) and some are thorny (e.g. *Acacia* spp., *Robinia pseudoacacia* and *Gleditsia triacanthos*), so the method is not effective for those. Some species cause photodermatitis (e.g. *Heracleum mantegazzianum*), so should be handled with care. Grazing is not possible to apply in forest canopy in selected EU countries.

Most harmful IAPS that can be managed

Heracleum mantegazzianum, Impatiens glandulifera, Lupinus polyphyllus, Oxalis pes-caprae (possible), *Robinia pseudoacacia* (mostly for regulating resprouts) and *Solidago* spp.

Grazing goats © ILRI, (CC BY-NC-ND 2.0) via Flickr

4.1.8. Clipping generative organs

General description

Clipping is usually based on manual removal of generative organs, mostly flowers before seed set. If species are vigorous, then repeated clipping is necessary to avoid seed dispersal. In cases of clonal plants, blocking the generative reproduction may induce vegetative resprouting.

Good implementation practices

The removed material needs to be destroyed (burnt).

Effectiveness

For monocarpic plants (e.g. *H. mantegazzianum*), it is an efficient method. The method is efficient for small infestations of non-clonal herbaceous species. This method is applicable in cases when the regular management action is delayed, so removing generative organs helps to prevent seeds being spread.

Costs, effort and resources required

There is no special equipment needed for clipping. Garden scissors or removing (hand-picking) is the most frequent method. Clipped flowers or seed pods need to be stored in plastic bags if transfered to a waste incinerator, or in paper bags if burned at a site by the managers' themselves.

Seed pods of *Asclepias syriaca* can be clipped to reduce the risk of seed dispersal © Jan Pergl

Potential side effects

Not known.

Feasibility and potential shortcomings

This method is very time-consuming and it only hinders generative reproduction, so it cannot serve as a long-term solution for many species.

Most harmful IAPS that can be managed

Asclepias syriaca, Buddleja davidii, Heracleum mantegazzianum and *Lupinus polyphyllus*

4.1.9. Root cutting

General description

Root cutting is based on destroying the upper part of the root of non-vegetatively reproducing herbaceous species. Roots must be cut at least 10–15 cm below the ground, at the beginning of the growing season (before flowering, April to June) and left on the ground to become dry. Similar to root cutting is the so-called "spring digging", which is done early in the spring using a hoe when the plants emerge (end of March–April). This is important, especially in the case of *H. mantegazzianum*, when the plants are small, the root is only 5-10 cm deep and the risk of contact with leaves (and burning) is small (Pergl 2017).

Good implementation practices

On wet sites the roots need to be removed from the site or placed on the foliage of the up-rooted plants without soil contact.

Effectiveness

Very effective method suitable for relatively small areas or low density stands of IAPS.

Costs, effort and resources required

No special resources are needed. Only the labour, and digging and protection equipment (e.g. gloves) are required.

Potential side effects

Not known.

Feasibility and potential shortcomings

The method is suitable for species like *H. mantegazzianum* and can be applied to areas with relatively low coverage, and areas up to approximately 500 m^2.

Most harmful IAPS that can be managed

Heracleum mantegazzianum and *Lupinus polyphyllus*

Digging up the roots of a *Heracleum* plant © Jan Pergl

4.1.10. Mowing and mulching

General description

Mowing is preferred over mulching, as it reduces the amount of nutrients in the system. It should be noted, however, that many species react to mowing by developing short aboveground biomass with small inflorescences. Therefore, the height of the following cuts needs to be adjusted appropriately. For species that respond to mowing by growing in the ground (e.g. *Ambrosia* spp. and *Heracleum mantegazzianum*), the mowing height should be gradually reduced, because mowing repeatedly at the usual height will allow them to avoid further destruction and to form seeds.

If the locality is infested with species capable of vegetative reproduction, then the mowing method can be used temporarily, but must eventually be replaced by herbicide application to completely eradicate the species. If there are no seeds or propagules, the harvested biomass can be freely used, but the possibility of regeneration has to be taken into account.

Good implementation practices

Timing of the mowing/mulching is crucial and it has to be done before flowering and setting seeds. If there is a risk of any seeds ripening from non-mature seeds, the cut biomass must be harvested immediately and not left *in situ.*

Effectiveness

Mowing and grazing are usually only complementary management measures to reduce seed production of IAPS. Mowing and mulching are often applied in large areas of infestation, or in sites with restrictions on the use of herbicides (e.g. organic farms, protected areas). Mowing and mulching are usually not efficient methods for eradication, but they can be suitable for the long-term management of populations and depletion of the seed bank.

Costs, effort and resources required

Suitable methods for large scale management of IAPS by regular agricultural mechanisation. Several mowing efforts per season are needed due to the regeneration ability of many species.

Potential side effects

Mowing will not eliminate the population and may initiate stand thickening. The mechanisation can be a vector of dispersal of propagules from invaded sites. Therefore, cleaning the equipment is essential.

Feasibility and potential shortcomings

Mowing and mulching are applicable at various stands, including large and homogeneous stands where machinery can be used. At small and heterogeneous sites, manual equipment is usually used. Mulching can be preferred at these sites, to reduce the costs of biomass transport. However, mulching can hide invasion foci and make the following management problematic.

Most harmful IAPS that can be managed

Heracleum mantegazzianum, Impatiens glandulifera, Lupinus polyphyllus, Oxalis pes-caprae and *Solidago* spp.

4.1.11. Biocontrol

General description

Biocontrol is a technique that uses naturally-occurring host-specific insects, mites or pathogens to help control invasive alien species. Biocontrol can be very cost-effective, though it does not result in full eradication. This approach is limited to those weeds for which safe biological control agents have been identified, tested, and authorised. It is important to note that biocontrol is not currently regulated at the EU level, however any application for its use needs to be done in accordance with national legislation. Before any release of an alien species as a biological control agent, an appropriate thorough risk assessment should be made.

In the context of European IAPS, the most promising species for which biocontrol might be used are *Acacia saligna*, water plants such as water hyacinth, water lettuce and *Ailanthus altissima* (Lesieur et al. 2023). Additionally, for *Opuntia* spp., effective biocontrol agents already exist, which have been used in various regions of the world, although not in Europe (see Novoa et al. 2019 for a list and further details).

For *A. altissima*, a highly specialised agent called Ailantex, based on the fungus *Verticillium nonalfalfae* (Maschek & Halmschlager 2018, Lechner et al. 2023), which causes *Ailanthus* wilting, is in some countries already used (Austria) or tested for its management and effectivity (Czech Republic). A disadvantage is the relatively long decay (2-3 years) of *Ailanthus* stands. Also, Ailantex is not biocontrol in strict sense, as the fungus has to be applied to each clone and does not spread spontaneously. Additionally, for *Acacia saligna*, there is a biocontrol program using bud-galling wasps successfully running in Portugal (Marchante et al. 2017). Results from biocontrol of knotweeds in the UK are not clearly positive and the agent has not fully set in the invaded areas (Shaw et al. 2009). The UK has also been testing the use of biocontrol against *Impatiens glandulifera*, but the results are not yet fully available (Pollard et al. 2021).

Good implementation practices

The best way of implementing biocontrol largely depends on the biocontrol agent. The spread and establishment of the agent needs to follow specific guidelines to assure its effectivity.

Effectiveness

If available as an option, the method can be effective. Even if not eradicating the IAPS fully, it should reduce their abundance.

Costs, effort and resources required

Research, testing, implementation and monitoring of biocontrol agents is costly and time consuming. However, in the long-term, it can be cheaper, or a better investment of resources, compared to the overall costs of other classical management methods.

Potential side effects

There is the risk of the biocontrol agent switching target organism to a native species. However, these risks can be mitigated through the application of rigorous evidence based risk assessments and testing (Sheppard et al. 2019).

Feasibility and potential shortcomings

In any case of using biocontrol, national legislation needs to be followed. If the method is approved, then e.g. in case of *Ailanthus altissima*, the application of Ailantex is straightforward and can be applied to large polycormons, which can be hard to manage by other methods.

Most harmful IAPS that can be managed

Opuntia spp.

The effects of biocontrol on an *Acacia* tree © Jan Pergl

Galls forming on the branches of an *Acacia* tree as a result of biocontrol © Jan Pergl

4.2. General considerations for all manual/mechanical methods

The timing of any type of intervention is crucial; if it is performed too early, the plants have time to regenerate and form seeds, whereas with late interventions, germinating seeds are often already formed before the management action. Management usually needs to be repeated within a growing season, so that seed formation does not occur during the season. It is important to always manage the entire stand to prevent regeneration from any remaining individuals. Management of neighbouring areas must also be ensured, e.g. to limit further seed supply from surrounding areas. This is especially important when large scale disturbances occur due to the management actions put in place.

Ice plant (*Carpobrotus edulis*) removal. Invasive species control through public participation, LIFE Biodiscoveries (LIFE13/BIO/PT/000386) © 2020 Lúcia Santos. All rights reserved. Licenced to the European Union under conditions.

1. Disposing of the material - composting

To prevent the establishment of IAPS at new sites, management requires careful handling of biomass from managed species and soil contaminated with rhizomes or seeds. Many species are able to regenerate even from small fragments of rhizomes and stems or from seeds persisting in the soil for several years. For species with a rhizome system or a perennial seed bank, soil should be deposited separately during manipulation to prevent spread over the whole area. At sites of nature conservation value and limited nutrient availability, it is necessary to ensure that all biomass is removed, so that decomposing material does not enrich the soil, especially with nitrogen, which slows down the recovery of natural communities (e.g. *Lupinus* spp., *Robinia* spp.). In areas where elevated nitrogen content is not a risk, biomass can be left and mulched. For species at risk of regeneration from above-ground biomass, the biomass should be crushed. Biomass left *in situ* must not contain rhizomes and seeds, or pollinated flowers that could produce germinating seeds. To limit the transport of biomass and the possible dispersal of seeds and rhizome fragments into the surrounding area, it is possible to collect the uprooted plants on an impermeable plastic and then remove/spray any regenerating plants with herbicide. The plastic must be removed before the winter period.

It is also possible to use the removed biomass in biogas plants or industrial composting areas. If the biomass contains seeds or vegetative parts capable of regeneration, this use is only possible if the heat treatment is high and long enough to destroy them. The treatment of biomass of IAPS must be prioritised to avoid storage and possible regeneration. Composting in home composters is not recommended due to the unstable temperature that may not be sufficient to dispose of the seeds. Burning is an appropriate method of disposal of dry biomass, and is recommended for the disposal of biomass of flowering or fruiting plants, parts of plants or seeds themselves, and also for the disposal of dried rhizomes. The advantage of on-site burning is that the risk of unintentional transport of propagules is minimised.

2. Monitoring

Monitoring of the management actions (e.g. success rate, costs incurred) must include checking the field work being carried out, and conformity with the methodologies and timetables. The effectiveness of the interventions must be assessed after actions are completed, but also for several years thereafter (at least 5 years), and the management actions be repeated, if necessary.

4.3. Summary overview of non-chemical management methods

Table 4: Summary overview of all non-chemical management methods, their effectiveness, resources needed, side effects and suitability for different IAPS.

Method	Effectiveness	Effort and resources required	Side effects	Not suitable for	Most harmful IAPS suitable to be managed*
Covering soil with plastic sheets	Effective for annuals and seedlings	Cheap method for small areas	Needs revegetation; risk of damage of the plastic	Perennial or rhizomatous species	*Carpobrotus* spp., *H. mantegazzianum, I. glandulifera, O. pes-caprae*
Pulling	Effective for annuals and young seedlings of perennials; sandy habitats	Cheap method, demanding on labour and training	No side effects; if done incorrectly, risk of quick regeneration	Adult perennials	Seedlings *B. davidii, R. ponticum; Carpobrotus* spp., *I. glandulifera, L. polyphyllus, Opuntia* spp., *O. pes-caprae, Solidago* spp.
Heat treatment	Effective for sensitive life stages (seedlings)	Costly; needs energy supply	Risk of fire	Seeds and plants with roots	*Carpobrotus* spp., *H. mantegazzianum, I. glandulifera, L. polyphyllus, Oxalis* sp., *Solidago* spp.
Girdling	Limited due to regeneration	Demanding on time, skills and mechanisation	Risk of poor management and rapid regeneration	Long term management	*R. ponticum, R. pseudoacacia*
Cutting	Effective for non-resprouting species	Cheap	None, if done properly on the right species	Resprouting species	
Digging up the underground organs	Effective for large trees, herbaceous species with tap root, sandy habitats, shrubs managed by large mechanisation	Costly mechanisation, cleaning or depositing the soil	Risk of unintentional spread; need to deal with the contaminated soil	Rhizomatous species with large polycormons (*Fallopia* spp.)	*B. davidii, Carpobrotus* spp., *H. mantegazzianum, L. polyphyllus, Opuntia* spp., *O. pes-caprae, R. ponticum, R. pseudoacacia, Solidago* spp.

* Note: Please see sections above for examples of more species that can be managed using these methods.

Method	Effectiveness	Effort and resources required	Side effects	Not suitable for	Most harmful IAPS suitable to be managed*
Grazing	Effective for non-resprouting IAPS and reducing the impact of resprouting IAPS, long term management, large areas	Need for complementary management (mowing, etc.)	Risk of transporting seeds and propagules with animals; adding nitrogen to the soil	Rapid eradication, toxic plants, habitats with complicated access	*H. mantegazzianum, I. glandulifera, L. polyphyllus, Solidago* spp.
Clipping generative organs - flowers	Effective for preventing seed' production and further spread	Demanding on time and labour	None	Large plants (shrubs and trees)	*A. syriaca, B. davidii, H. mantegazzianum, L. polyphyllus*
Root cutting	Effective for herbaceous species with tap root	Demanding on time and labour	None	Large areas, woody roots, rhizomatous species	*H. mantegazzianum, L. polyphyllus*
Mowing and mulching	Effective for non-resprouting IAPS and reducing the impact of resprouting IAPS, long term management, large areas	Part of traditional land use; suitable for large areas	Risk of transporting seeds and propagules; mulching changes the community composition	Rhizomatous species (*A. syriaca*, *Fallopia* spp.)	*H. mantegazzianum, I. glandulifera, L. polyphyllus, O. pes-caprae, Solidago* spp.
Biocontrol	Long term effectiveness; suitable for large polycormons or populations that cannot be managed mechanically	Precise application; delay of results due to long process	None	Species for which there is a lack of biocontrol agents	Potential to use for *Opuntia* spp.

* Note: Please see sections above for examples of more species that can be managed using these methods.

4.4. Notes on using chemical management methods

Integrated application of chemical methods alongside manual/mechanical methods is often used in order to ensure the highest management effectiveness of IAPS. A combination of manual/mechanical and chemical methods is often recommended when the former cannot be used independently due to leading to regeneration and vegetative rejuvenation of species, which causes thickening and expansion of their cover. Even though the use of herbicides is sometimes not the preferred option, in many cases their application leads to more effective eradication of IAPS, which is also associated with less needed visits to the invaded site, less disturbance, etc. (Pergl et al. 2020, Hocking et al. 2023). If possible, selective herbicides with direct stem injection are preferred, as well as stem debarking and application of herbicide on the scar or by using a herbicide torch. The injection of herbicide into the trunk (into drilled holes) or on the wound after bark stripping are referred to as targeted application methods. These are environmentally friendly methods, as they have

Drill holes filled with herbicide. Control and eradication of the invasive exotic plant species *Ailanthus altissima* in the Alta Murgia National Park, LIFE Alta Murgia (LIFE12/BIO/IT/000213) © 2020 Francesca Casella. All rights reserved. Licenced to the European Union under conditions.

Young individuals of *Ailanthus altissima* or other resprouting trees and shrubs can be treated by herbicide to debarked stem. It is important to debark only ca. 60% of the diameter of the stem. © Jan Pergl

no side effects when used correctly (as opposed to foliar spraying), so can be applied in sensitive and valuable natural sites, such as protected areas. Their main advantage is their high efficiency, as the vast majority of individuals die off after a single application without producing resprouts. The method is applicable for trees and shrubs and requires that trees are left to die spontaneously, after which they can be left to decay or completely dead trees can be felled. Foliar spraying of herbicides should be seen as a last option for large areas and guidelines for each herbicide need to be followed. When any form of chemical control is planned, it needs to always be done according to the relevant local and national legislation, and to the National Action Plans for the Sustainable Use of Plant Protection Products.

Herbicides are effective for the management of most IAPS. However, even using chemical management methods, for some rhizomatous species, especially when the management targets old and developped stands of IAPS, the treatment needs to be repeated and, for others, even after several years individuals may not be completely eradicated (e.g. *Fallopia* spp.).

5. Factsheets for the terrestrial invasive alien plant species most impactful to pollinators in the EU

5.1. *Asclepias syriaca*, Common milkweed

Native range	Non-native range	Pathways of introduction and spread
North America	Europe. Present in most Member States, widespread and common in the central, eastern and Southeastern regions, especially in the Pannonian Basin.	Introduced to Europe in the 17th century for ornamental purposes. Spreads through natural spread of seeds by wind, often along roads and railways, and through anthropogenic spread of seeds and rhizome fragments with machinery and soil (Follak et al. 2021). Popular with beekepers.

Note: The species is listed as of Union concern since 2017.

Common milkweed (*Asclepias syriaca*) © Ryan Hodnett, (CC BY-SA 4.0) via Wikimedia Commons

Biology and ecology

Asclepias syriaca is a perennial herb that reproduces both by seeds and rhizomes. The seeds are easily spread by wind over long distances and survive in the soil for more than five years. The shoots are annual and die back in autumn, with new shoots growing from the rhizome system each spring. The rhizomes usually grow horizontally at a depth of 10-40 cm but can reach up to 1.5 m deep.

Impacts on pollination

Bagi (2008) reported that the species can detract bees from pollinating sunflowers, and cause crop loss. Szigeti et al. (2020) found that honeybees and bumblebees showed a preference for milkweed above native plants; overall however, they did not find negative effects on pollinator communities. Kovács-Hostyánszki et al. (2022) found no significant differences between control and invaded sites at the plant species level, but found a significantly reduced abundance of hoverflies at *A. syriaca* invaded sites. *Asclepias syriaca* preferably colonises sand dune habitats, which offer suitable habitat for many wild bee species, including endemic and threatened species. Grasslands are very important habitats for pollinating insects (Kudrnovsky et al. 2020) and decreases of native grassland species (Kelemen et al. 2016) may have negative effects on specialist pollinating insects. Negative impacts on non-pollinating invertebrates have been demonstrated (e.g. Galle et al. 2015, Somogyi et al. 2017, Kapilkumar et al. 2019).

Management

Asclepias syriaca is able to regenerate well from rhizome fragments. This poses a risk, e.g. during soil transport or by contamination of agricultural machinery. In order to achieve complete eradication of *A. syriaca*, only chemical methods (foliar spraying of large stands and direct application of herbicides on leaves e.g. by herbicide torch) are currently known to be effective (Table 5).

In areas where herbicide cannot be used, seed formation must be prevented by manual/mechanical methods. However, this will not lead to eradication, but will instead stimulate branching and further spread, since it responds to damage by vigorous vegetative growth and by branching from underground rhizomes. However, it is possible to manually clip the seed pods before their opening, though it is important to note that the sap of *A. syriaca* is toxic, so gloves and protective clothes are needed.

Common milkweed (*Asclepias syriaca*) © Agnieszka Kwiecień, Nova, (CC BY-SA 4.0) via Wikimedia Commons

Mowing and grazing are only complementary management measures to reduce seed production, which can be used temporarily, but if complete eradication is needed these methods must eventually be replaced by herbicide application. Mowing will not eliminate the population and may initiate stand thickening. Two mowing efforts per season are advisable - the first just before flowering (at the flower bud stage) and the second before regenerating individuals' flowers. Mowing at an earlier stage of development induces faster regeneration, requiring frequent repetition of the intervention. The biomass must always be harvested and disposed of in an appropriate manner. Grazing is less effective than mowing because the sward is not sought after by animals (it can cause health problems) and areas not grazed need to be controlled. If grazed areas produce fruit, there is a risk of seeds being spread on the animals' coats and, in addition, seeds being incorporated into the soil by their hooves.

Digging up the roots is very laborious and does not lead to the complete eradication of the species on the site. Ploughing is only marginally effective (only effective on seedlings), but it can stimulate regeneration. Moreover, both methods carry the risk of further spreading the species through plant material left on machinery or in contaminated soil. Pulling can only be used effectively on young seedlings that do not have a branched rhizome system.

Table 5: List of methods used for management of *Asclepias syriaca*, indicating their suitability for different management objectives, potential unintended effects and relevant notes. Methods that are not recommended to be used to manage the species are shown at the end.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Clipping generative organs (seed pods)			Effective for stopping spread	Human labour demanding, not leading to eradication of the population
Pulling/Digging up the underground organs	Only for juvenile plants (seedlings)	Not recommended	Not recommended	May induce regeneration
Ploughing	Marginally effective (only effective on seedlings)	Not recommended	Not recommended	Stimulates branching and further spread; risk of unintentional spread
Herbicides	Effective	Effective	Effective	Total herbicides are more efficient than selective ones
Grazing	Not recommended	Not recommended	Not recommended	May be toxic for animals
Mowing and mulching	Not recommended	Not recommended	Not recommended	Stimulate branching and further spread; risk of unintentional spread

Recommended literature regarding management of the species:

- Bakacsy & Bagi (2020)
- Berki et al. (2023)
- Csiszár & Korda (2017)
- Follak et al. (2021)
- Lapin (2017)

Asclepias syriaca © Ruslan Kastani, (CC BY-SA 4.0) via Wikimedia Commons

5.2. *Buddleja davidii*, Butterfly bush

Native range	Non-native range	Pathways of introduction and spread
Temperate Asia (China)	Europe, North America, South America, South Africa, Australia, New Zealand	Introduced to Europe in the late 19th century for ornamental purposes. Spreads through natural spread of seeds by wind and water, often along roads and railways, and mostly anthropogenic spread of seeds (CABI 2009a).

Biology and ecology

Buddleja davidii is a perennial, semi-deciduous shrub or small multi-stemmed tree originating from China. In central Europe, it invades disturbed locations like gravel river banks. This shrub typically reaches a height of 3-5 m. Its fragrant flowers bloom in racemose inflorescences, measuring 10-25 cm long, with lilac-coloured corollas. The fruit is a capsule. There are around 50 known varieties and cultivars. *Buddleja davidii* plants readily reproduce asexually from stem and root fragments, and individuals that are disturbed by flooding and mechanical means have been observed regenerating from buried stems, stumps, and roots soon after the disturbances. Debris from *B. davidii* left on floodplains can also regenerate, flower, and spread if left on site. Regarding its seedbank, *B. davidii* has a relatively short seed viability. In laboratory conditions, seed viability remains high for up to 2.5 years, but declines rapidly between 2.5 and 3.5 years.

Impacts on pollination

The large flowers attract generalist butterflies (via olfactory cues, Lehner et al. 2022), wasps, hornets, lacewings, beetles and honeybees (Tallent-Halsell & Watt 2009); nectar is provided, but the plant is not used by any European butterfly species as the host for larval development. Pollinators might be withdrawn from native plants (Giuliano et al. 2004, Corcos et al. 2020) in natural habitats, as it can form monodominant stands, prevent growth and displace pioneer native species and accelerate succession (CABI 2009a, Tallent-Halsell & Watt 2009).

Butterfly bush (*Buddleja davidii*) © Julia Sumangil, (CC BY-SA 4.0) via Wikimedia Commons

Management

Controlling *B. davidii* using solely manual/mechanical methods has shown mixed results. To limit its spread, it is recommended to remove seed capsules (dead-heading, clipping) before they ripen. On a small scale, physically removing young shrubs (cutting or digging up the whole plants with roots) can help in the early stages of invasion, but it is not ideal for mature plants in well-established populations. For cut plants, the application of glyphosate herbicides (always in accordance with local, national and EU legislation) is known to be effective. Larger shrubs with dense pubescent leaves are less susceptible to foliar application with herbicide, therefore precise methods like targeted application on stumps or drilling and injection are effective, but require more effort (Table 6).

Buddleja davidii debris is problematic because stem and root fragments can easily regrow. Piles of debris that are not properly dealt with (burned, composted, or treated to kill all seeds and fragments) can become a concentrated source of new plants in the next season. Once *B. davidii* establishes itself in a disturbed area, it becomes challenging to remove or manage. Manual removal is labour-intensive and costly, while herbicides work in small areas in the short term but require repeated manual application.

Table 6: List of methods used for management of *Buddleja davidii* indicating their suitability for different management objectives, potential unintended effects and relevant notes.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling	Only for juvenile plants (seedlings)	Not recommended	Not recommended	
Cutting	Not recommended (regeneration)	Not recommended (regeneration)	Effective to reduce the seed set	Stimulates branching and further spread; need to properly deal with the biomass
Digging up the underground organs	Possible	Possible	Possible	Need to deal properly with the soil (seeds) and debris
Clipping generative organs (seed pods)			Effective for stopping spread	Human labour demanding at large scales
Herbicides	Effective	Effective	Effective	Application on stumps, injection to drilled holes or on removed bark

Recommended literature regarding management of the species:

- Tallent-Halsell & Watt (2009)

5.3. *Carpobrotus edulis, C. acinaciformis* and their hybrids, Ice plant

Native range	Non-native range	Pathways of introduction and spread
South Africa	North America, South America, North Africa, Australia, New Zealand, Europe (on coastal habitats of southern and western Europe)	Introduced for ornamental purposes, and for soil and sand dune stabilisation. Spread through natural spread of seeds by birds and mammals and shoot fragments; anthropogenic spread both intentionally and unintentionally.

Note: For the controversial taxonomy of *C. edulis* and *C. acinaciformis* and their hybrids see e.g., Campoy et al. (2018) and Novoa et al. (2023).

Ice plant (*Carpobrotus edulis*) © R~P~M, (CC BY-NC-ND 2.0) via Flickr

Biology and ecology

Carpobrotus spp. are notable invasive species which are problematic mainly in the Mediterranean region due to their aggressive spread. They are succulent perennial plants with thick, fleshy leaves that range in colour from bright green to bluish green. The flowers are large and daisy-like, from pink to purple to yellow in colour. It rapidly forms dense mats and outcompetes native vegetation through aggressive growth. The species are able to reproduce through seeds, and vegetatively through trailing stems and stem fragments. They occupy various habitats, including coastal areas, rocky shores, sand dunes, and disturbed sites.

Impacts on pollination

Carpobrotus spp. are pollinated by Hymenoptera (bumblebees, honeybees and wild bees). Competition for pollinators with native plant species, but also neutral and facilitative effects, depending on the species, have been found (Moragues & Traveset 2005, Suehs et al. 2005, Jakobsson et al. 2008, Bartomeus et al. 2008b, Vilá et al. 2009); as well as reduced seed production in a native species (Jakobsson et al. 2008). It is a potential threat to the conservation of many endemic Mediterranean plant species in coastal areas and sand dunes and therefore to their pollinators, which in some cases are endemic and highly species-specific (León et al. 2023).

Management

Controlling *Carpobrotus* spp. is challenging and is usually done by physical removal (e.g. digging/pulling), solarisation, potentially prescribed fire and chemical treatment (Table 7). However, these approaches may have limitations, such as being labour intensive, costly, or potentially harmful to native species. Effective management often requires long-term efforts to prevent re-establishment. A review of management actions done in Europe is available in Chenot et al. (2018) and Campoy et al. (2018).

Physical removal can be highly effective, particularly in sandy soils where it requires less physical effort. However, this method can be time consuming when dealing with extensive *Carpobrotus* spp. infestations. For such cases, the most efficient approach involves rolling up the *Carpobrotus* spp. mat from one side while severing the roots beneath using shovels. Alternatively, a brush rake can also prove effective. Once *Carpobrotus* spp. have been manually removed, it is crucial to transport the plant material to a secure location to dry. If possible, covering it with black plastic can expedite the drying process and prevent root and fragment regeneration. However, managing and disposing of the plant material can present logistical challenges.

Ice plant (*Carpobrotus edulis*) © Sandy Thomas, (CC BY-NC 4.0) via iNaturalist

Ice plant (*Carpobrotus edulis*) © Ed Suominen, (CC BY-NC 2.0) via Flickr

Solarisation entails covering the soil with plastic sheets to control smaller *Carpobrotus* spp. infestations. However, it is not recommended for extensive infestations, as it can lead to significant and lasting physical, chemical, and biological changes in the soil. Some studies have found solarisation to be ineffective in managing *Carpobrotus* spp. invasions.

Prescribed fire can potentially be used to control *Carpobrotus* spp., as the high temperatures during a controlled burn can effectively kill *Carpobrotus* spp. seeds stored in the topsoil. However, due to many possible unintended effects and lack of experience, using controlled burning in areas with *Carpobrotus* spp. infestations should be approached with careful consideration.

Chemical control, primarily through herbicides, proves most effective when dealing with pure *Carpobrotus* spp. clumps. Additionally, chemical control by spraying may be suitable after manually removing *Carpobrotus* spp. to prevent regrowth from seeds.

Removal of *Carpobrotus* species. Island conservation in Tuscany, restoring habitat not only for birds, RESTO CON LIFE (LIFE13/NAT/IT/000471) © 2019 Michele Giunti. All rights reserved. Licenced to the European Union under conditions.

Table 7: List of methods used for management of *Carpobrotus* spp. indicating their suitability for different management objectives, potential unintended effects and relevant notes.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Digging up the underground organs /Pulling	Recommended	Recommended	Effective	Rolling up the *Carpobrotus* spp. mat from one side while severing the roots beneath using shovels
Covering soil with plastic sheets	Effective	Not recommended for large areas	Effective	
Heat treatment (prescribed fire)	No clear efficiency, eradicates seeds	No clear efficiency, eradicates seeds	Effective	
Herbicides	Effective	Effective	Effective	Herbicide is applied by using foliar spraying

Recommended literature regarding management of the species:

- Chenot et al. (2018)
- Campoy et al. (2018)

5.4. *Heracleum mantegazzianum*, Giant hogweed

Native range	Non-native range	Pathways of introduction and spread
Caucasus	North America, South America, Australia, New Zealand, Europe (widespread)	Introduced to Europe in 1890 for ornamental purposes, as pasture for bees (honey plant), and a cover plant (for hunting). Spreads mainly through spread of wind dispersed seeds and water.

Note: The species is listed as of Union concern since 2017.

Biology and ecology

Heracleum mantegazzianum is a monocarpic (dies after flowering) species and reproduces exclusively through seeds. It begins flowering at the age of 3-5 years, but in unsuitable conditions the plant may wait until it is 12 years to do so. The species forms a short persistent seed bank, with the majority of seeds germinating within the first and second years. However, a small proportion of seeds can persist in the soil for up to 7 years, posing a risk of re-infestation.

Heracleum mantegazzianum © Katrin Schneider, (CC BY-SA 4.0) via Wikimedia Commons

Giant hogweed (*Heracleum mantegazzianum*) © ChristianWernerZH, (CC BY-SA 2.0) via Flickr

Impacts on pollination

Heracleum mantegazzianum outcompetes and replaces native plant species, which may lead to changed plant communities and shifts in plant-depending pollinators. Zumkier (2012) compared the pollinator community (visitation) and plant fitness (seeds set of co-flowering plant) of *H. mantegazzianum* and the native *H. sphondylium* in an experimental garden setup and found low competition between the two species, a neutral effect on seed set, and a high attractiveness of the large *H. mantegazzianum* inflorescences for honeybees. Davis et al. (2018) found lower abundances of solitary bees and hoverflies at invaded compared to uninvaded sites. Grace and Nelsen (1981) did not find much overlap of flower visitors between *H. mantegazzianum* and the native *H. sphondylium* and Nielsen et al. (2008) found little evidence of negative effects on pollination. Recently, Bogusch et al. (2023) found that, while the flowers of *H. mantegazzianum* were frequently visited by high abundances of insects, the community's pollinators were relatively species poor and uneven, with a few generalist Diptera species and the honeybee dominating over all other flower visitors. They concluded that giant hogweed is not a necessary part of flower communities for flower visiting insects, and it should be eradicated because of its negative effects on other plants, landscape and humans.

Management

The preferred mechanical method to manage this species is root cutting, where it is crucial to cut it at least 10 cm below the ground. In areas with silty soil or long-grazed areas, deeper cutting may be necessary. The excavated roots can either be left to dry on the surface or, in waterlogged sites, should be removed. The method is suitable in areas with low cover and smaller populations (up to approximately 500 m² and around 200 individuals) (Table 8). Pulling the seedlings is not advised due to high risk of removing only leaves and following regeneration from roots.

As *H. mantegazzianum* is a monocarpic species, a suitable approach is to target flowering plants in small stands. To prevent seed production, plants should be prevented from setting seeds by removing the inflorescences. Timing is crucial; the intervention should occur when plants are in full bloom or at the start of fruit formation (typically June to July). Care must be taken to avoid seed release during handling and repeated visits are needed.

Mowing and grazing are effective for controlling large stands and depleting the seed bank when herbicide use is limited. These methods do not kill plants, but delay flowering and lower seed production. Grazing can be carried out by animals such as sheep or cattle, but caution is needed as some animals can be sensitive to the sap. Regardless of the chosen method, management should be repeated to ensure any unaffected or overlooked individuals are addressed, because *H. mantegazzianum* has a high regeneration potential. Maintenance should continue for at least 5-10 years to deplete the seed bank effectively. Managing neighbouring areas is necessary to prevent the spread back to mown or grazed areas.

Another method involves using a dense plastic cover placed over the invaded sites in February-March before mass germination. This cover, made from dense plastic and securely fixed to the ground, causes all hogweed plants underneath to die. The cover should be removed before winter. Hot treatments (foam, water) can be used for managing young seedlings.

Heracleum mantegazzianum is sensitive to a wide range of herbicides which can be applied directly to leaves by spraying or by injection.

Field workers scything large Giant hogweed (*Heracleum mantegazzianum*). Awareness building, surveying and controlling invasive alien species (IAS) in Finland, Finvasive LIFE (LIFE17/NAT/FI/0528)

Table 8: List of methods used for management of *Heracleum mantegazzianum* indicating their suitability for different management objectives, potential unintended effects and relevant notes. Methods that are not recommended to be used to manage the species are shown at the end.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Covering soil with plastic sheets	Effective	Not recommended	Not recommended	Suitable for small areas, plastic must be removed; revegetation actions needed
Heat treatment (water/foam)	Effective for young seedlings	Not recommended	Not useful for flowering plants	Suitable for very young seedlings and small areas
Ploughing	Not recommended	Not recommended	Not effective due to regeneration	Not recommended due to high regeneration and seeds germinated from seed bank
Grazing	Not recommended	Reduces the seed set, not killing the plants	Effective, but need to repeat	Need to repeat, suitable for large areas
Clipping generative organs (flowers)	Recommended	Not recommended (human labour demanding)	Effective	
Root cutting	Recommended	Not recommended (human labour demanding)	Effective	
Mowing	Not recommended	Reduces the seed set, not killing the plants	Effective, but need to repeat	Need to repeat, suitable for large areas
Herbicides	Effective	Effective	Effective	No need to use herbicides for adult plants, effective for seedlings
Pulling	Not recommended	Not recommended	Not effective	

Recommended literature regarding management of the species:

- Nielsen et al. (2005)
- Pergl (2019)
- Pyšek et al. (2007)
- Rajmis et al. (2016)
- Tiley et al. (1996)

Mown giant hogweed stands must be repeatedly managed, as cut plants quickly and easily regenerate © Jan Pergl

5.5. *Impatiens glandulifera*, Himalayan balsam

Native range	Non-native range	Pathways of introduction and spread
Himalayas	North America, South America, Asia, New Zealand, Europe (present in almost all Member States and biogeographic regions, where it is particularly abundant in the Atlantic and Continental regions)	Introduced to Europe in 1839 for ornamental purposes. It was also introduced as pasture for bees (honey plant). Spreads via contaminated soil and garden waste; seeds can be transported with rivers over long distances.

Note: The species is listed as of Union concern since 2017.

Biology and ecology

Impatiens glandulifera is an annual plant reaching heights of up to 2.5 meters. This plant reproduces only by seeds and if damaged, it can regenerate from the nodes. It invades primarily riparian habitats, but can also flourish in damp woodlands and waste grounds.

Impacts on pollination

It forms dense stands that cover the soil, and shade out and replace native annual and perennial plant species because of early germination and rapid growth. *Impatiens glandulifera* is nectar-rich and its flowers attract more pollinators, especially bumblebees, than native plants, having a negative effect on the fitness of the native plant species through competition for pollinators, luring pollinators away from native species (Prowse & Goodridge 2000, Chittka & Schürkens 2001). Lopezaraiza-Mikel et al. (2007) and Nielsen et al. (2008) found that *I. glandulifera* invaded plots had significantly higher visitor species richness, visitor abundance and flower visitation. However, this did not translate in facilitation for pollination, as more generalised insects were more likely to visit the alien plant. Nienhuis et al. (2009) found highest proportion of visitors for *Bombus* spp. in *I. glandulifera* invaded sites and for solitary bees in sites where *I. glandulifera* had been removed, while no negative impacts on general pollinator abundance or functional insect diversity were found. Bartomeus et al. (2010) found no evidence that *I. glandulifera* outcompetes native plants for pollinators, and that pollinator abundances depend on landscape structure, but are modulated by this mass- and late-seasonal floral resource. They also found *I. glandulifera* receiving higher visitation rates than simultaneously flowering native plants, mainly from bumblebees, but no differences of visitation rates for the plant community, except

Himalayan balsam (*Impatiens glandulifera*) © Thomas Bresson, (CC BY 2.0) via Flickr

for the honeybee which increased their visits in invaded sites. Cawoy et al. (2012) studied effects of *I. glandulifera* on visitation rate, insect behaviour, pollen deposition and reproductive success of two native plant species (*Epilobium angustifolium* and *Aconitum napellus* ssp. *lusitanicum*) and found that proximity and abundance increased bumblebee visitation rates, while abundance had a negative effect on honeybee visits to both native plant species. Bumblebees preferred *I. glandulifera* and deposited considerable quantities of alien pollen on the native plants, without significantly decreasing seed set. Also, Emer et al. (2015) found that the relationship between flower visitation and pollen load or deposition is not straightforward for *I. glandulifera* and pollen transfer networks are more complex. Recently, Najberek et al. (2023) found that increased bumblebee visitation to *I. glandulifera* increased transmission rates of pathogen fungi, which could pose a serious threat both to native biodiversity and nearby crop production.

Management

Management efforts of this species need to include a focus on preventing the spread to new areas, especially through soil transport contaminated with seeds. When complete eradication is the goal, the management must start upstream and progress throughout the watershed where the species is present.

A recommended method is the pulling up of entire plants, including their roots. Pulling should target mature or regenerating plants, ideally in the early flowering stage before the seed formation. Treated areas need regular inspection (approximately every 2-3 weeks) to pull any remaining individuals. Continued monitoring is necessary for several subsequent years. For local eradication efforts, removing all plants is essential, along with preventing seed influx into the area. Pulled plants can be left in place only if the intervention occurred before flowering and the plants do not contain any seeds. Measures should prevent re-rooting, such as separating roots from stems and breaking the uprooted plants to minimise regeneration and the development of adventitious roots from nodes (Table 9).

Mowing is a suitable method for managing extensive stands. In the first year, mowing should be done 2-3 times and the cut plants should either be trampled or mulched. Within about two seasons, the stand will reach a stage where individual pieces can be uprooted. Plants should be mowed as low as possible (below the first node) to limit regeneration. The first mowing should occur no later than the budding stage of the first flowers. Early mowing leads to plant regeneration, while later mowing allows production of seeds. Mowing should be repeated multiple times per season, and control areas should be inspected for emerging plants.

Grazing can lead to plant trampling, subsequent regeneration, and seed production. Thus, a combined approach is recommended. The vegetation should be mowed or mulched before flowering and then allowed for grazing, with simultaneous monitoring and removal of flowering plants.

Mowing and grazing can be used as landscape management tools to limit seed production. However, these methods should be executed with care to minimise seed spread to uninvaded areas. Given the existence of effective mechanical methods, herbicide application is not recommended.

Himalayan balsam (*Impatiens glandulifera*) © John Knight, (CC BY 2.0) via Flickr

Heat treatment by foam or water can be effective for young seedlings, as they are heat/frost sensitive in the beggining of the vegetation season. Repeated visits to the site are neccesary due to continuous germination.

Table 9: List of methods used for management of *Impatiens glandulifera* indicating their suitability for different management objectives, potential unintended effects and relevant notes.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling	Recommended	Recommended, effective	Recommended, effective	Need to carefully handle the biomass
Heat treatment (hot water/foam)	Possible	Not recommended	Not recommended	Can be used for seedlings
Grazing	Not recommended	Possible, recommended for large areas. Does not kill all plants, but can be used for lowering species density	Possible, recommended for large areas. Can be used for lowering species density and seed set	
Mowing	Not recommended	Possible, recommended for large areas. Does not kill all plants, but can be used for lowering species density	Possible, recommended for large areas. Can be used for lowering species density and seed set	
Herbicides	Not recommended	Not recommended	Not recommended	Not needed, mechanical methods are effective

Recommended literature regarding management of the species:

- Beerling & Perrins (1993)
- Helmisaari (2010)
- Karlovarský kraj (2015)
- Nienhuis et al. (2009)
- Oliver et al. (2020)
- Saegesser et al. (2016)
- Tanner (2017)

Himalayan balsam (*Impatiens glandulifera*) © Maja Dumat, (CC BY 2.0) via Flickr

5.6. *Lupinus polyphyllus*, Large-leaved lupine

Native range	Non-native range	Pathways of introduction and spread
North America	South America, Australia, New Zealand, Europe (widespread)	Introduced to Europe in 1829 for ornamental purposes. Has been introduced as pasture for bees (honey plant), for soil improvement (nitrogen addition) and stabilisation, and fodder for livestock and game animals. Spreads through seeds via contaminated soil and garden waste; vehicles and machinery (often along roads and railways); natural spread of seeds by water, livestock and wildlife.

Note: An EU-Risk Assessment was produced in 2016 (conclusion: High Risk with high confidence), but the species is not listed as of Union concern.

Biology and ecology

Lupinus polyphyllus is a perennial herbaceous plant from the Fabaceae family. It has high regeneration capacity after shoot damage and reproduces through seeds. The lifespan of the plant is under 20 years, and while some forms contain alkaloids that are mildly toxic, alkaloid-poor variants are used as fodder for wildlife and domestic animals. The impact of *L. polyphyllus* is mainly through enriching the soil with nitrogen.

Impacts on pollination

Lupinus polyphyllus reduces plant diversity and changes plant communities due to nitrogen enrichment in the soil and its toxic compounds (e.g. Hejda et al. 2009, Loydi et al. 2015), with subsequent effects on insect communities. Valtonen et al. (2006) found that plant species richness, cover and diversity along road verges in Finland was lower in *L. polyphyllus* invaded sites compared to non-invaded sites. The abundance of butterflies and diurnal moths was also lower in *L. polyphyllus* invaded sites. Jakobsson & Padrón (2014) found that *L. polyphyllus* attracted bumblebees and that other flower resources (potted *Lotus corniculatus*) became increasingly available for solitary wild bees. Jakobsson et al. (2015) found that *L. polyphyllus* facilitated pollination of two native plants (potted *Lotus corniculatus* and *Lychnis viscaria*), but only at the local scale (within 5 m). The bumblebees' preference for *L. polyphyllus* during the flowering period is of concern, as the plant diverts and detracts bumblebees from pollinating native plants.

Large-leaved lupine (*Lupinus polyphyllus*) © Joe Mabel, (CC BY-SA 2.0) via Wikimedia Commons

Management

Single *L. polyphyllus* plants or small stands can be mechanically removed by uprooting, but this has to be repeated several years and over the season to prevent seed set (Table 10).

The species benefits from irregular management. Therefore timely, regular, consistent, and long-term mowing has to be implemented. To prevent seed set, mowing should occur before seeds become mature; even green and soft seeds can still lead to plant regeneration. Regular mowing and grazing are important for conserving semi-natural grasslands. Additional mowing is often necessary when *L. polyphyllus* is flowering. All biomass must be removed to prevent nitrogen enrichment of the soil.

Recent studies have explored heat treatments to kill *L. polyphyllus* seeds, but their seeds have thick seed coats, making them resistant to heat treatments. Removing the flowers can be used to reduce the seed set. The plant is decorative, so the general public can be asked to collect the inflorescences. To be effective, the plants have to be managed repeatedly over the season, as they regenerate quickly. Removing seed pods is not recommended due to vertical ripening of the pods and risk of seed release.

Chemical control is effective for this species.

Large-leaved lupine (*Lupinus polyphyllus*) © George Chernilevsky, Public domain, via Wikimedia Commons

Table 10: List of methods used for management of *Lupinus polyphyllus* indicating their suitability for different management objectives, potential unintended effects and relevant notes. Methods that are not recommended to be used to manage the species are shown at the end.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling/Digging up the underground organs	Possible, only for small areas	Not recommended	Possible, only for small areas	Has to be repeated several years and over the season to prevent seed set
Grazing	Effective if repeated for several years	Effective if repeated for several years	Recommended	Effective for limiting seed set; does not kill the plants in short period
Clipping generative organs		Not recommended	Effective	Human labour demanding; does not kill the plants
Mowing	Effective if repeated for several years	Effective if repeated for several years	Recommended	Effective for limiting seed set; does not kill the plants
Herbicides	Effective	Effective	Effective	
Ploughing	Not recommended	Not recommended	Not recommended	Stimulates the regeneration from underground organs; risk of unintentional spread

Recommended literature regarding management of the species:

- Eckstein et al. (2023)
- Fremstad (2010)
- Ramula (2020)

Large-leaved lupine (*Lupinus polyphyllus*) © George Chernilevsky, Public domain, via Wikimedia Commons

5.7. *Opuntia* spp., Prickly pear

Native range	Non-native range	Pathways of introduction and spread
North and South America	Africa, Tropical Asia, Australia, Europe	Different *Opuntia* spp. have been introduced to Europe as early as the 1500s for ornamental purposes and for the dying industry. These species are used as fruit and fodder crop, as landscaping and as windbreaks. Natural spread of seeds occurs by animals and water. It can spread also by fragments of stem (paddlelike cladodes).

Note: Among the most commonly introduced species in the EU are *Opuntia ficus-indica*, *Opuntia stricta* and *Austrocylindropuntia* spp.

Biology and ecology

Opuntia spp. are a group of succulent plants native to the Americas. They are characterised by their flat, paddle-shaped stems called cladodes, which are covered in spines and glochids (tiny hair-like structures). They produce colourful, often edible fruits and showy flowers. They are adapted to various climates, from dry arid deserts to more temperate regions, and are known for their resilience and ability to thrive in diverse ecosystems. They are often introduced as ornamentals, food and forage, dye and as medicinal plants.

The primary method of spreading for opuntioid cacti is vegetative. This form of dispersal occurs throughout the year when cladodes, immature fruit, or flowers detach and come into contact with the ground. Small cladodes or parts of them from many opuntioid cacti easily attach themselves to clothing, footwear, and the fur and limbs of animals, which aids in their dispersal. Additionally, new growth and spread can also occur from the flowers, giving rise to new segments or roots.

Prickly pear (*Opuntia* spp.) © John Winder, (CC BY-NC-ND 2.0) via Flickr

Impacts on pollination

Opuntia spp. grow in dense, monospecific stands, displace native and endemic plant species (and invertebrates, Robertson et al. 2011) and modify natural succession. *Opuntia* spp. are mostly pollinated by medium- and large-sized bees (Tenorio-Escandón et al. 2022). Bartomeus et al. (2008a) found that *Opuntia* spp. competed for pollinators with native species and affected the plant-pollinator network structure by significantly attracting more pollinator species, and having more pollinator visitations than native plants in the area. Padrón et al. (2009) and Vilà et al. (2009) found that *Opuntia* spp. modified the number of links between plants and pollinators via integration of the most generalist pollinators, but did not affect native network properties.

Management

During control activities, it is essential to eliminate all cladodes and fruits from the area, since they have the capacity to regenerate, even under severe conditions. Under appropriate conditions, physical control methods, such as manual (by hand) or mechanical (using machinery) removal, can present a practical and cost-effective approach for managing all opuntioid cactus species. Hand removal proves highly effective for small plants and scattered infestations, while machinery-based removal works well for large, densely packed, and impenetrable stands. However, it is important to note that physical removal can dislodge cladodes, which have the potential to regrow into new plants. Proper disposal is crucial and should involve deep burial (Table 11).

In terms of biocontrol of opuntioid cacti, two main agents are employed: *Cactoblastis cactorum*, a stem-boring moth, and various *Dactylopius* species, cochineal scale insects. *Cactoblastis cactorum* has been highly effective in controlling *O. stricta* (common prickly pear) in most situations. Establishing infections in vulnerable cacti species with *C. cactorum* and cochineal insects can be relatively straightforward, either by introducing the agents as eggs (*C. cactorum*) or by placing infected segments onto unaffected plants (*C. cactorum* larvae or cochineal *Dactylopius*). Biocontrol is the most cost-effective option for managing widespread cactus invasions and has been successful in various regions of South Africa, USA, Australia and Spain. More details about *Opuntia* biocontrol can be found in Novoa et al. (2019).

Opuntia spp. also respond positively to herbicide stem injection techniques, including pad injection and drill-and-fill methods.

Table 11: List of methods used for management of *Opuntia* spp. indicating their suitability for different management objectives, potential unintended effects and relevant notes.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling/ Digging up the underground organs	Effective	Effective	Effective	Potential to regrow from waste
Biocontrol	Effective	Recommended	Recommended	
Herbicides	Effective	Effective	Effective	Stem injection techniques

Recommended literature regarding management of the species:

- Potter & Sheehan (2017)
- Novoa et al. (2019)

Prickly pear (*Opuntia* spp.) © John Tann, (CC BY 2.0) via Flickr

5.8. *Oxalis pes-caprae*, Sourgrass

Native range	Non-native range	Pathways of introduction and spread
South Africa	North America, South America, North Africa, Asia, Australia, New Zealand, Europe (mostly in the Mediterranean)	Introduced to Europe at the end of the 18th or the beginning of the 19th century for ornamental purposes. Spreads through unintentional and natural vegetative spread by seeds and bulbils that are dispersed by soil movement and garden waste, small mammals, vehicles and machinery, and water.

Biology and ecology

Oxalis pes-caprae is a perennial herb with underground bulbs, a height of ca. 10-30 cm, and with tufted habitus. It grows in the vicinity of human settlements, on the edges of paths, as a weed in vegetable crops, in thickets, and at the edges of forests. It is usually found in slightly shaded places, but also occurs in sunny, sandy and rocky sites. It flowers from December to May.

Sourgrass (*Oxalis pes-caprae*) © Jörg Hempel, (CC BY-SA 2.0) via Flickr

Impacts on pollination

Jakobsson et al. (2009) and Albrecht et al. (2016) found spatial scale-dependent effects of the alien plant on the pollination of native plants. In the latter study, at the landscape scale, honeybees and bumblebees where attracted into invaded sites, translating into native plant visitation facilitation by honeybees, while bumblebees almost exclusively visited the non-native *O. pes-caprae.*

Management

The preferred management method for small infestations is hand pulling, which can be applied to plants of all sizes. In more compacted substrates, hand pulling must be made during the rainy season, as to facilitate the removal of the bulbs. Training of the workers is needed, as it is important to also remove bulbs, so no bulbs are left in the ground. The hand pulling method must be repeated several times over the year to weaken the plant (Table 12).

The effects of mowing, grazing and cutting the species are not known. Based on the ecology of the species, the effectiveness of these methods is assumed to be low (due to the presence of tubers) and their use is associated with the risk of unintentional transport of seeds, tubers and contaminated soil.

It is very important to limit the risk of spreading bulbs due to the transport or natural movement of contaminated soil. It is also important to adopt adequate cleaning techniques of any machinery that operates in infested areas. In agricultural infested areas, typical agronomic preventive and control measures, such as crops rotation, should be adopted.

Biocontrol through parasitic plants (*Orobanche* spp.) has been considered, but the effectiveness is unclear.

Foliar application of herbicides is effective.

Table 12: List of methods used for management of *Oxalis pes-caprae* indicating their suitability at different scale, context and effectiveness management objectives, potential unintended effects and relevant notes.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling	Low effectiveness	Recommended, but demanding on time at large infestations with many bulbils	Recommended	Need to remove whole plants, including the bulbils
Grazing	Low effectiveness	Low effectiveness	Possible	The species is palatable, but may cause sheep intoxication in case of excessive ingestion
Mowing	Low effectiveness	Low effectiveness	Possible	
Herbicides	Effective	Effective	Effective	Broad spectrum herbicides are more efficient than selective ones

Recommended literature regarding management of the species:

- Marshall (1987)
- Lazzaro et al. (2019)

Sourgrass (*Oxalis pes-caprae*) © Dr. Alexey Yakovlev, (CC BY-SA 2.0) via Flickr

5.9. *Rhododendron ponticum*, Pontic rhododendron

Native range	Non-native range	Pathways of introduction and spread
Iberia, Caucasus	Western Europe, Asia	Introduced to Great Britain in 1763 and to Germany in 1784 for ornamental purposes. Planted as a windbreak, used for erosion control, and as a cover plant (for hunting). Spreads through natural spread of seeds by wind and water.

Biology and ecology

Rhododendron ponticum is an evergreen shrub introduced as a cultivated flowering plant. The seeds from *R. ponticum* ripen in December and dispersal begins in the beginning of spring. Seeds are very small and are designed for dispersal primarily by wind. Seeds remain viable for up to one year. Mature plants that are already established on suitable sites expand their area of occupation either by vegetative spread through stem layering or by seed dispersal and seedling establishment.

Pontic rhododendron (*Rhododendron ponticum*) © vm56, (CC BY-NC) via iNaturalist

Impacts on pollination

Rhododendron ponticum contains grayanotoxins, which are secondary compounds in the nectar that are neurotoxic to honeybees and some wildbees and beneficial to some bumblebees (Tiedeken et al. 2014, 2016). *Rhododendron ponticum* causes changes to the composition of the flower-visiting community (Dietzsch et al. 2011, Stout & Casey 2014, Tiedeken & Stout 2015, Tiedeken et al. 2016) by replacing native species.

Management

Seedlings and relatively young small bushes can be pulled by hand, which can be used to eradicate small scale invasions. Ideally, early detection will allow management before the plants can reproduce (ca. <10 years old) (Table 13).

For larger more established invasions, integrated management is needed, using both physical and chemical measures, the choice and application being determined by the different stages of the invasion (i.e., plant growth). See IUCN (2019a) which summarises the best practices set out in Edwards (2006) and Higgins (2008) to manage this species. Stem treatment is the most effective and efficient method of killing large *R. ponticum* bushes. Where

there is no access to treat the stems, an overall foliar spray is the next option. Another option for larger bushes is to reduce their size by cutting and then applying the herbicide. Without herbicide application, *R. ponticum* can regenerate easily from cut or flailed stumps.

At sites where the use of herbicides is not possible, partial girdling (ca. 70% of perimeter) is recommended. The method causes slow exhausting of the adult tree with reduced resprouting. This method is recommended above the high stump, but it needs to leave the tree to die out several years.

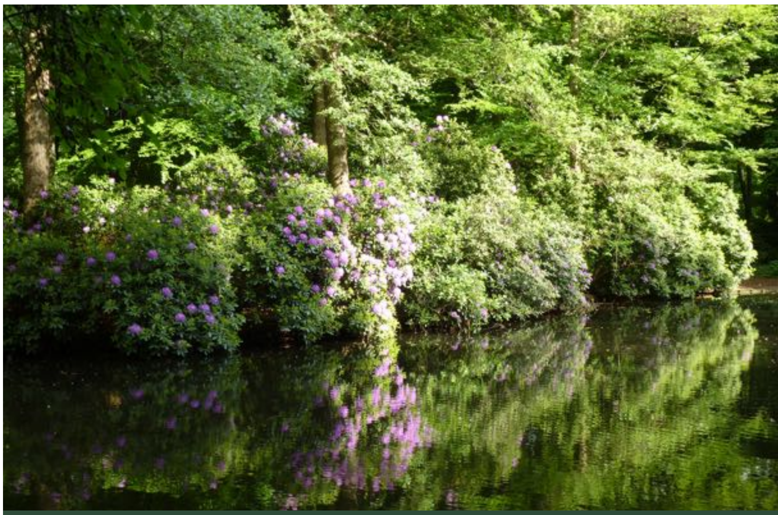
Rhododendron ponticum © Krzysztof Ziarnek, (CC BY-SA 4.0) via Wikimedia Commons

Table 13: List of methods used for management of *Rhododendron. ponticum* indicating their suitability for different management objectives, potential unintended effects and relevant notes.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling/Digging up the underground organs	Effective only for juvenile plants	Effective only for juvenile plants	Effective	
Partial girdling	Not effective enough	Not effective enough	Reduces the seed set (but may induce large vegetative lateral spread)	Has to be done in several years (see method description), not in sunny and open habitats
Cutting	Not effective (high regeneration)	May be used for enabling access for herbicide application	Effective	Rapidly regenerating species
Herbicides	Effective	Effective	Effective	Stump application or foliar spraying

Recommended literature regarding management of the species:

- IUCN (2019a)
- Edwards (2006)
- Higgins (2008)



Rhododendron ponticum © Katrin Schneider, (CC BY-SA 4.0) via Wikimedia Commons

5.10. *Robinia pseudoacacia*, Black locust

Native range	Non-native range	Pathways of introduction and spread
North America	North America, South America, Africa, Asia, New Zealand, Europe (widespread)	Introduced to Europe between 1623 and 1635 for ornamental purposes. Has been lately largely planted for forestry, erosion control and pasture for bees (´honey plant´). Reproduces through seeds that can be spread via contaminated soil and garden waste, vehicles and machinery and water. Regenerates rapidly by root suckers when damaged.

Biology and ecology

Robinia pseudoacacia is a deciduous broadleaf tree that typically grows to a height of 12 to 18 m, but can reach heights of up to 30 m under favourable conditions. It has a curved trunk and a sparse, irregular crown. On warm, rocky slopes, it exhibits a shrub-like growth pattern, reaching heights of 3 to 5 m, with an untamed crown and twisted trunk. It belongs to the Fabaceae family and thus is able to add nitrogen to the soil.

Impacts on pollination

The negative impacts of *R. pseudoacacia* on biodiversity include competition with native plants, nitrogen enrichment in the soil, and succession and habitat modification with subsequent changes at higher trophic levels (insect and bird communities) (e.g. Buchholz et al. 2015, Reif et al. 2016, Poblador et al. 2019). Buchholz and Kowarik (2019) tested whether the attractiveness of *R. pseudoacacia* vs. a common native plant (*Cytisus scoparius*) for pollinators changed with increasing urbanisation and found higher visitation rates for *R. pseudoacacia*, but no significant differences in actual flower access. *Robinia pseudoacacia* visits decreased with increasing urbanisation, but the authors consider *R. pseudoacacia* still a “pollinator-friendly” tree for certain urban settings.

Black locust (*Robinia pseudoacacia*) © Agnieszka Kwiecień, Nova, (CC BY-SA 4.0) via Wikimedia Commons

Black locust (*Robinia pseudoacacia*) © Agnieszka Kwiecień, Nova, (CC BY-SA 4.0) via Wikimedia Commons

Management

Robinia pseudoacacia management must consider the species' high sprouting ability, which is stimulated by damage, even in older individuals. After cutting *R. pseudoacacia*, vigorous regeneration occurs within a radius of up to 15 meters from the removed individual, making clear-cutting or strip cutting highly ineffective (Table 14). *Robinia pseudoacacia* has a preference for light, thus it does not spread well into closed forest stands. Seedlings of *R. pseudoacacia* thrive only on disturbed bare soil. Seeds of *R. pseudoacacia* do not germinate well and do not disperse to large distances.

In commercial forests with scattered *R. pseudoacacia* occurrences, clear-cutting should be avoided. Due to *R. pseudoacacia*'s light preference, it should be replaced with selective cutting and promoting natural stand regeneration by releasing native species to create canopies and allow for fruiting. Incremental thinning should progressively favour younger individuals or groups of target trees. On steppic sites, the sprouting of *R. pseudoacacia* can be limited by sheep and goat grazing. Goats are preferred for long-term care, as they actively seek *R. pseudoacacia* leaves and shoots. Trees in cities can be dug out by mechanisation, but the resprouting shoots need to be treated by herbicide. *Robinia pseudoacacia* management requires subsequent treatment over 3-5 years.

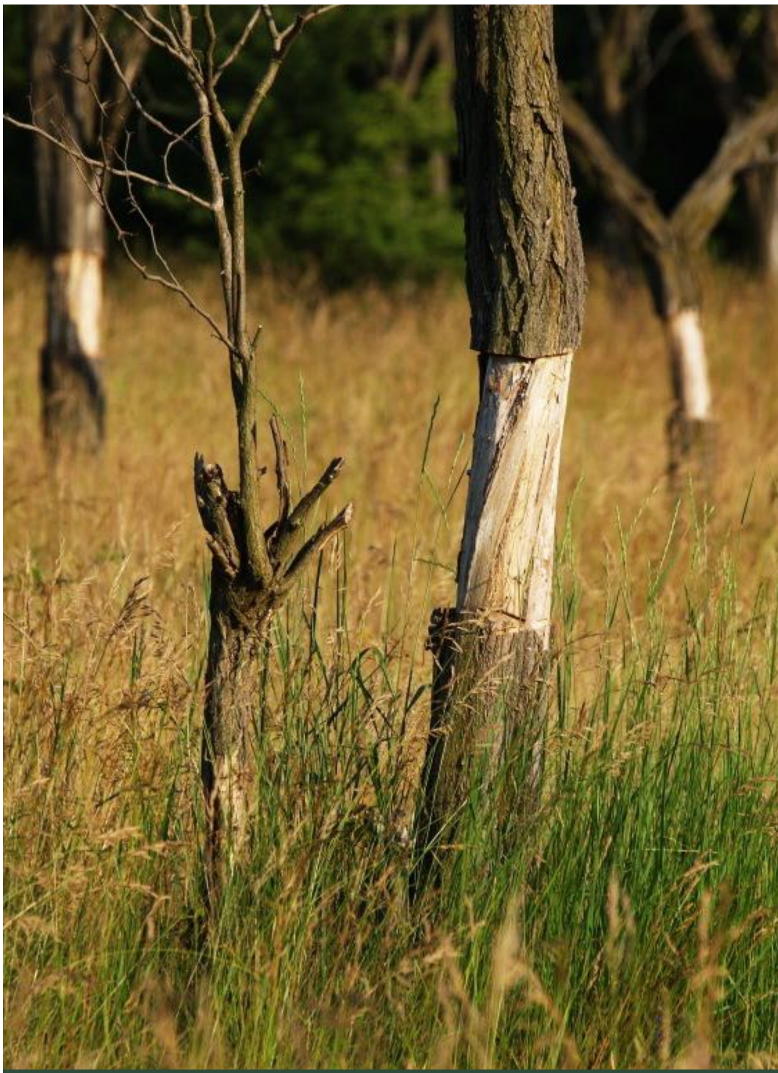
Local eradication cannot be achieved by using only mechanical control. Effective management strategies involve targeted herbicide application methods directly to the tree. In the case of young individuals, herbicide is applied to the wound after partial bark removal. These targeted application methods require that the treated trees are subsequently left untouched for spontaneous decay. In exceptional cases, where trees cannot be left standing for the necessary herbicide absorption period, they can be cut to a high stump. After the emergence of shoots from the stump, herbicide can be injected into the stump and the root suckers treated. However, this approach is more time-consuming, requires specific materials, and demands high-quality subsequent treatment.

Other methods, such as cutting to a high stump with delayed injection or cutting to a low stump with immediate herbicide application to the cut surface, are less effective compared to targeted herbicide application to the intact trunk. These methods result in the formation of new shoots, necessitating further herbicide use for their control.

At sites where the use of herbicides is not possible, partial girdling (ca. 70% of perimeter) is recommended. The method causes slow exhausting of the adult tree with reduced resprouting. This method is recommended above the high stump, but it needs to leave the tree to die out several years.

In the case of excessively dense and continuous stands, foliar spraying is necessary, which carries the risk of damaging the surrounding vegetation. The resulting bare surface is susceptible to recolonisation by undesirable species.

On nutrient-poor sites, all biomass from treated individuals must be removed to prevent the enrichment of the soil with nitrogen, which slows down the recovery of natural communities. The harvested area should not be ploughed, as soil disturbance significantly promotes *R. pseudoacacia* regeneration. If biomass burning is practiced, it should not occur in areas where *R. pseudoacacia* is present, as the species is also stimulated by fire (root regeneration and seed germination).



Black locust (*Robinia pseudoacacia*) © HermannFalkner/sokol, (CC BY-NC 2.0) via Flickr

Table 14: List of methods used for management of *Robinia pseudoacacia* indicating their suitability for different management objectives, potential unintended effects and relevant notes. Methods that are not recommended to be used to manage the species are shown at the end.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling	Can be effective for small seedlings	Not recommended	Not recommended	Induces regeneration
Partial girdling	Not effective enough	Not effective enough	Reduces the seed set (but may induce large vegetative lateral spread)	Has to be done in several years (see method description), not in sunny and open habitats
Digging up the underground organs	Individual trees can be dug out	Not recommended		If not properly dug, may induce regeneration from remaining roots
Grazing	Does not kill the tree, can be used for regulating resprouts	Does not kill the tree, can be used for regulating resprouts	Reduces the seed set (but may induce large vegetative lateral spread)	
Herbicides	Effective	Effective	Effective	
Cutting	Not effective, must be combined with herbicide application; tall stumps show relatively smaller regeneration, but the cutting does not kill the tree	Not effective, must be combined with herbicide application; tall stumps show relatively smaller regeneration, but the cutting does not kill the tree	Not effective, cutting induce strong vegetative spread	Induces massive regeneration and resprouting

Recommended literature regarding management of the species:

- Csiszár & Korda (2017)
- Sádlo et al. (2017)

Drilling holes to the stem, with application of herbicides, is an effective method for eradicating *Robinia pseudoacacia* © Jan Pergl

5.11. *Solidago* spp. (*S. canadensis* and *S. gigantea*), Canadian and Giant goldenrod

Native range	Non-native range	Pathways of introduction and spread
North America	Asia, Australia, New Zealand, Europe (widespread)	Introduced to Europe in the 17th (*S. canadensis*) and 18th (*S. gigantea*) century for ornamental purposes and also introduced as pasture for bees (´honey plant´). Spread through unintentional spread of rhizomes and seeds via contaminated soil and garden waste, vehicles and machinery (often along roads and railways), animals, and artificial corridors (water canals). Natural spread occurs through seeds by wind.

Solidago canadensis © Marek Slusarczyk, (CC BY 3.0) via Wikimedia Commons

Biology and ecology

Solidago canadensis and *S. gigantea* are perennial clonal herbaceous plants with small, wind-dispersed seeds. The species establish themselves quickly from wind-dispersed seeds and start flowering in their second year. *Solidago* spp. also spread clonally, forming monodominant stands. The species can be found spreading vigorously along transportation routes, roadsides, and in disturbed areas including construction sites.

Impacts on pollination

Negative effect on the diversity and abundance of wild pollinators, regardless of their nesting and food specialisation (Moron et al. 2009). Reduction in native plant species richness and negative effects on the abundance of bees, but not of hoverflies, have been reported. Native flowers experienced reduced visitation by wild bees, honeybees and hoverflies due to the augmented presence of *S. canadensis* (Fenesi et al. 2015). Native plant species richness and the richness, abundance and diversity of butterfly species were lower in invaded stands of *S. canadensis*. Hoverfly abundance, diversity and species richness were negatively affected in July before the onset of flowering of *S. canadensis*, but tended to be positively affected in August during the height of flowering (De Groot et al. 2007).

Management

The species thrives under irregular management practices, such as mowing followed by several years of no intervention. Recommended measures to control the spread of these species include repeated mowing (for several years). Small, localised populations of *Solidago* spp. can be uprooted by hand pulling; however, the risks of disturbing the vegetation should be considered. In the case of persistent populations that do not respond adequately to mechanical management, herbicide application (foliar spraying) can be considered according to national legislation (Table 15).

Ploughing the sites invaded by *Solidago* spp. is not recommended, as it stimulates regeneration, fragments the rhizomes, disturbs the soil and allows its colonisation by seeds. Mowing or grazing are mechanical methods that can reduce the seed set.

Canadian goldenrod (*Solidago canadensis*) © Forest and Kim Starr, (CC BY 2.0) via Flickr

Table 15: List of methods used for management of *Solidago* spp. indicating their suitability for different management objectives, potential unintended effects and relevant notes. Methods that are not recommended to be used to manage the species are shown at the end.

Method	Rapid eradication/manag. small populations	Management of established sites/large populations	Control spread	Unintended effects and notes
Pulling	Effective if repeated for several years	Not recommended, human labour demanding	Effective	
Grazing	Effective if repeated for several years	Effective if repeated for several years	Effective	
Mowing	Effective if repeated for several years	Effective if repeated for several years	Effective	
Herbicides	Effective	Effective	Effective	Foliar spraying
Ploughing	Not recommended	Not recommended	Not recommended	Stimulates regeneration; risk of unintentional spread

Recommended literature regarding management of the species:

- Csiszár & Korda (2017)
- IUCN (2019b)

Canadian goldenrod (*Solidago canadensis*) © F. D. Richards, (CC BY-SA 2.0) via Flickr

6. Conclusions and potential limitations

This report briefly summarises information on terrestrial IAPS in the EU having a negative impact on pollination, with a focus on the non-chemical methods that can be used to manage them. Around 50 publications since 2000 were analysed and relevant IAPS prioritised based on their distribution and impact on pollination in Europe.

Regarding the IAPS that currently present the highest threat to pollinators in the EU, the number of European occurrence records included in GBIF only has illustrative character, as the data were not thoroughly checked for specific details on the species distribution. In addition, the literature search performed was not exhaustive, so it is possible that some information on other species or impacts has been missed. A more informed prioritisation of species should ideally be based on more robust data or by using an expert-consensus exercise.

It is evident, however, that there is a lack of specific studies on impacts of IAPS on pollinators and pollination. Arguably, the most harmful – and not yet regulated in the EU – species are the two goldenrods, *Solidago canadensis* and *Solidago gigantea*. Note that *Solidago altissima*, another North American species, has been recorded only recently from Europe for the first time, at a single location in Belgium (Verloove et al. 2017). As such, these three species would be good candidates for developing full risk assessments and for potential future listing under the EU IAS Regulation.

Regarding the use of non-chemical options to manage IAPS, mechanical methods are usually sufficiently effective for managing annual species (e.g. *Impatiens glandulifera*) or can be used for non-resprouting perennial species (e.g. *Heracleum mantegazzianum*). For perennial species with clonal dispersal ability, mechanical methods often do not lead to eradication of stands, even after several years of repeated application. What is more dangerous, applying solely mechanical methods can induce regrowth and worsen the situation. The same limitations of these methods can also apply to relatively small annual species creating large populations with high density (e.g. *Senecio inaequidens*). The use of large machinery such as bulldozers and backhoes can be effective, as it removes whole plants and soil with seeds, but it does result in the need to store and/or treat large amounts of contaminated soil. In the case of plants that reproduce by seeds, a wide range of mechanical methods may be used to prevent seed production. Nevertheless, it is important to carry out management methods with caution and to minimise the movement of biomass and contaminated soil, and to always ensure that machinery is cleaned.

In areas where herbicides cannot or should not be used, seed formation must be prevented by mechanical methods. However, this will not kill the stands of many invasive species (e.g. *Asclepias syriaca*); on the contrary, it will stimulate their growth and spread, since those species respond to damage by vigorous vegetative growth and by branching from underground rhizomes and roots, or by activation of adventive buds. Thus, for many resprouting species (e.g. *Asclepias syriaca*, *Robinia pseudoacacia*), purely mechanical methods are inappropriate or not effective enough and should be combined with targeted herbicides application (if habitat conditions allow). Biocontrol, which has been successfully tested and applied for some of the species, offers another option in the management toolbox, but it is essential to carefully assess potential risks and side effects.

This short and preliminary overview of IAPS with negative impacts on pollinators in the EU, and the possible non-chemical management options to reduce these impacts, has demonstrated that the available information is fragmented. Some of the species with the highest impact are very widely distributed in Europe and management will be challenging, but rewarding, even if only at the local or regional scale. In a future horizon scanning exercise for IAPS, special attention should be given to possible impacts on pollinators, to support preventive measures. The impacts of IAPS (and invasive alien animal species) on pollinators and pollination provide a unique opportunity to counteract more than one important environmental problem with appropriate and targeted measures and fulfil obligations provided by the EU Pollinators Initiative and the EU Regulation on Invasive Alien Species. In addition, this report highlights only a few of the problematic IAPS in Europe, and therefore can be taken as a guideline for species having similar life-forms, invaded habitats, etc. Much of the information shown here is relevant for other species such as *Ailanthus altissima* (very similar to *Robinia pseudoacacia*), *Senecio inaquidens* (counterpart of *Carpobrotus edulis*, but only pulling and herbicides can be recommended), and *Fallopia* spp. (partly similar to *Asclepias syriaca*).

7. References

Adler, L.S. (2001): The ecological significance of toxic nectar. Oikos 91: 409-420.

Albrecht, M., Padrón, B., Bartomeus, I. & Traveset, A. (2014): Consequences of plant invasions on compartmentalization and species' roles in plant–pollinator networks. Proceedings of the Royal Society B: Biological Sciences 281: 20140773.

Albrecht, M., Ramis, M.R. & Traveset, A. (2016): Pollinator-mediated impacts of alien invasive plants on the pollination of native plants: the role of spatial scale and distinct behaviour among pollinator guilds. Biological Invasions 18: 1801-1812.

Alhmedi, A., Haubruge, E. & Francis, F. (2010): Intraguild interactions and aphid predators: biological efficiency of *Harmonia axyridis* and *Episyrphus balteatus*. Journal of Applied Entomology 134: 34-44.

Antoń, S., Komoń-Janczara, E. & Denisow, B. (2017): Floral nectary, nectar production dynamics and chemical composition in five nocturnal *Oenothera* species (Onagraceae) in relation to floral visitors. Planta 246: 1051-1067.

Arceo-Gómez, G. & Ashman, T.-L. (2016): Invasion status and phylogenetic relatedness predict cost of heterospecific pollen receipt: implications for native biodiversity decline. Journal of Ecology 104: 1003-1008.

Arnold, S.E.J., Idrovo, M.E.P., Arias, L.J.L., Belmain, S.R. & Stevenson, P.C. (2014): Herbivore defense compounds occur in pollen and reduce bumblebee colony fitness. Journal of Chemical Ecology 40: 878-881.

Auge, H. & Brandl, R. (1997): Seedling recruitment in the invasive clonal shrub, *Mahonia aquifolium* Pursh (Nutt.). Oecologia 110: 205-211.

Bagi, I. (2008): Common milkweed (*Asclepias syriaca* L.). In: Botta-Dukat, Z. & Balogh, L. (eds) The most important invasive plants in Hungary. HAS Institute of Ecology and Botany, Vacratot, 151-158.

Bajwa, A.A., Sadia, S., Ali, H.H., Jabran, K., Peerzada, A.M. & Chauhan, B.S. (2016): Biology and management of two important *Conyza* weeds: a global review. Environmental Science and Pollution Research 23: 24694-24710.

Bakacsy, L. & Bagi, I. (2020): Survival and regeneration ability of clonal common milkweed (*Asclepias syriaca* L.) after a single herbicide treatment in natural open sand grasslands. Scientific Reports 10: 1-10.

Baracchi, D., Marples, A., Jenkins, A.J., Leitch, A.R. & Chittka, L. (2017): Nicotine in floral nectar pharmacologically influences bumblebee learning of floral features. Scientific Reports 7: 1951.

Barov, B., Stoev, P., Sapundzhieva, A., Metodiev, T. & Hristova, K. (2022): Workshop Report 27. January 2022 Expert consultation workshop on identifying measures to tackle pressures from invasive alien species on wild pollinators. 18 pp.

Bartomeus, I., Bosch, J. & Vilà, M. (2008b): High invasive pollen transfer, yet low deposition on native stigmas in a *Carpobrotus*-invaded community. Annals of Botany 102: 417-424.

Bartomeus, I., Vilà, M. & Santamaría, L. (2008a): Contrasting effects of invasive plants in plant–pollinator networks. Oecologia 155: 761-770.

Bartomeus, I., Vilà, M. & Steffan-Dewenter, I. (2010): Combined effects of *Impatiens glandulifera* invasion and landscape structure on native plant pollination. Journal of Ecology 98: 440-450.

Baude, M., Leloup, J., Suchail, S., Allard, B., Benest, D., Mériguet, J., Nunan, N., Dajoz, I. & Raynaud, X. (2011): Litter inputs and plant interactions affect nectar sugar content. Journal of Ecology 99: 828-837.

Beerling, D.J. & Perrins, J.M. (1993): *Impatiens glandulifera* Royle (*Impatiens roylei* Walp.). Biological Flora of British Isles. No. 177. Journal of Ecology 81: 367-382.

Berki, B., Botta-Dukát, Z., Csákvári, E., Gyalus, A., Halassy, M., Mártonffy, A., Rédei, T. & Csecserits, A. (2023):

Short-term effects of the control of the invasive plant *Asclepias syriaca*: Secondary invasion of other neophytes instead of recovery of the native species. Applied Vegetation Science 26: e12707.

Bezemer, T.M., Harvey, J.A. & Cronin, J.T. (2014): Response of native insect communities to invasive plants. Annual Review of Entomology 59: 119-141.

Bjerknes, L., Totland, O., Hegland, S.J. & Nielsen, A. (2007): Do alien plant invasions really affect pollination success in native plant species? Biological Conservation 138: 1-12.

Bogusch, P., Vojtová, T. & Hadrava, J. (2023): High abundance but low diversity of floral visitors on invasive *Heracleum mantegazzianum* (Apiaceae). NeoBiota 86: 193-207.

Buchholz, S. & Kowarik, I. (2019): Urbanisation modulates plant-pollinator interactions in invasive vs. native plant species. Scientific Reports 9: 6375.

Buchholz, S., Tietze, H., Kowarik, I. & Schirmel, J. (2015): Effects of a major tree invader on urban woodland arthropods. PLoS One 10: e0137723.

CABI (2009a): Datasheet *Buddleja davidii* (butterfly bush). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.10314 (accessed 3.8.2023)

CABI (2009b): Datasheet *Cortaderia selloana* (pampas grass). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.11872 (accessed 3.8.2023)

CABI (2010): Datasheet *Mahonia aquifolium* (Oregongrape). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.32269 (accessed 3.8.2023)

CABI (2013a): Datasheet *Nicotiana glauca* (tree tobacco). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.36324 (accessed 3.8.2023)

CABI (2013b): Datasheet *Fallopia japonica* (Japanese knotweed). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.23875 (accessed 3.8.2023)

CABI (2014a): Datasheet *Arundo donax* (giant reed). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.1940 (accessed 3.8.2023)

CABI (2014b): Datasheet *Galinsoga parviflora* (gallant soldier) https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.25299 (accessed 3.8.2023)

CABI (2014c): Datasheet *Nicotiana tabacum* (tobacco). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.36326 (accessed 3.8.2023)

CABI (2014d): Datasheet *Senecio inaequidens* (South African ragwort). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.49557 (accessed 3.8.2023)

CABI (2017): Datasheet *Prosopis juliflora* (mesquite). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.43942 (accessed 3.8.2023)

CABI (2018): Datasheet *Elaeagnus angustifolia* (Russian olive). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.20717 (accessed 1.8.2023)

CABI (2019): Datasheet *Baccharis halimifolia* (groundsel-bush). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.8164#sec-6 (accessed 3.8.2023)

CABI (2021): Datasheet *Conyza bonariensis* (hairy fleabane). https://www.cabidigitallibrary.org/doi/10.1079/cabicompendium.15250 (accessed 3.8.2023)

Campoy, J.G., Acosta, A.T.R., Affre, L., Barreiro, R., Brundu, G., Buisson, E., González, L., Lema, M., Novoa, A., Retuerto, R., Roiloa, S.R. & Fagúndez, J. (2018): Monographs of invasive plants in Europe: *Carpobrotus.* Botany Letters 165: 440-475.

Cawoy, V., Jonard, M., Mayer, C. & Jacquemart, A. (2012): Do abundance and proximity of the alien *Impatiens glandulifera* affect pollination and reproductive success of two sympatric co-flowering native species? Journal of Pollination Ecology 10: 130-139.

Charlebois, J.A. & Sargent, R.D. (2017): No consistent pollinator-mediated impacts of alien plants on natives. Ecology Letters 20: 1479-1490.

Chenot, J., Affre, L., Gros, R., Dubois, L., Malecki, S., Passetti, A., Aboucaya, A. & Buisson, E. (2018): Eradication of invasive *Carpobrotus* sp.: effects on soil and vegetation. Restoration Ecology 26: 106-113.

Chittka, L. & Schürkens, S. (2001): Successful invasion of a floral market. Nature 411: 653-655.

COM (2021): 261 final. Report from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Progress in the implementation of the EU Pollinators Initiative. 16 pp.

COM (2023): Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Revision of the EU Pollinators Initiative. A new deal for pollinators. 20 pp.

Coombs, G. & Peter, C.I. (2010): The invasive 'mothcatcher' (*Araujia sericifera* Brot.; Asclepiadoideae) co-opts native honeybees as its primary pollinator in South Africa. AoB Plants 2010:plq021.

Corcos, D., Cappellari, A., Mei, M., Paniccia, D., Cerretti, P. & Marini, L. (2020): Contrasting effects of exotic plant invasions and managed honeybees on plant-flower visitor interactions. Diversity and Distributions 26: 1397-1408.

Csiszár, Á. & Korda, M. (eds) (2017): Practical Experiences in Invasive Alien Plant Control. Second revised and expanded edition. Rosalia Handbooks, Duna-Ipoly National Park Directorate, Budapest. 249 pp.

Davis, E.S., Kelly, R., Maggs, C.A. & Stout, J.C. (2018): Contrasting impacts of highly invasive plant species on flower-visiting insect communities. Biodiversity and Conservation 27: 2069-2085.

de Groot, M., Kleijn, D. & Jogan, N. (2007): Species groups occupying different trophic levels respond differently to the invasion of semi-natural vegetation by *Solidago canadensis*. Biological Conservation 136: 612-617.

Detzel, A. & Wink, M. (1993): Attraction, deterrence or intoxication of bees (*Apis mellifera*) by plant allelochemicals. Chemoecology 4: 8-18.

Dietzsch, A.C., Stanley, D.A. & Stout, J.C. (2011): Relative abundance of an invasive alien plant affects native pollination processes. Oecologia 167: 469-479.

DiTommaso, A. & Losey, J.E. (2003): Oviposition preference and larval performance of Monarch butterflies (*Danaus plexippus*) on two invasive swallowwort species. Entomologia Experimentalis et Applicata 108: 205-209.

Domènech, R., Vilá, M., Gesti, J. & Serrasolses, I. (2006): Neighbourhood association of *Cortaderia selloana* invasion, soil properties and plant community structure in Mediterranean coastal grasslands. Acta Oecologia 29: 171-177.

Eckstein, R.L., Welk, E., Klinger, Y.P., Lennartsson, T., Wissman, J., Ludewig, K., Hansen, W. & Ramula, S. (2023): Biological flora of Central Europe – *Lupinus polyphyllus* Lindley. Perspectives in Plant Ecology, Evolution and Systematics 58: 125715.

Edwards, C. (2006): Managing and controlling invasive rhododendron. Forestry Commission: Edinburgh https://cdn.forestresearch.gov.uk/2006/08/fcpg017.pdf

Emer, C., Vaughan, I.P., Hiscock, S. & Memmott, J. (2015): The impact of the invasive alien plant, *Impatiens glandulifera*, on pollen transfer networks. PLoS ONE 10: e0143532.

EPPO (2007): *Solanum elaeagnifolium.* EPPO Bulletin 37: 236-245.

EPPO (2018): Data sheet. *Prosopis juliflora* (Sw.) DC. EPPO Bulletin 49: 290-297.

EPPO (2023): EPPO Global Database (available online). https://gd.eppo.int.

Ernst, C. & Cappuccino, N. (2005): The effect of an invasive alien vine, *Vincetoxicum rossicum* (Asclepiadaceae), on arthropod populations in Ontario old fields. Biological Invasions 7: 417-425.

Fenesi, A., Vágási, C.I., Beldean, M., Földesi, R., Kolcsár, L.-P., Shapiro, J.T., Török, E. & Kovács-Hostyánszki, A. (2015): *Solidago canadensis* impacts on native plant and pollinator communities in different-aged old fields. Basic and Applied Ecology 16: 335-346.

Ferrero, V., Castro, S., Costa, J., Acuna, P., Navarro, L. & Loureiro, J. (2013): Effect of invader removal: pollinators stay but some native plants miss their new friend. Biological Invasions 15: 2347-2358.

Flanagan, R.J., Mitchell, R.J. & Karron, J.D. (2010): Increased relative abundance of an invasive competitor for pollination, *Lythrum salicaria*, reduces seed number in *Mimulus ringens*. Oecologia 164: 445-454.

Florentine, S.K. & Westbrooke, M.E. (2005): Invasion of the noxious weed *Nicotiana glauca* R. Graham after an episodic flooding event in the arid zone of Australia. Journal of Arid Environments 60: 531-545.

Follak, S., Bakacsy, L., Essl, F., Hochfellner, L., Lapin, K., Schwarz, M., Tokarska-Guzik, B. & Wołkowycki, D. (2021): Monograph of invasive plants in Europe N°6: *Asclepias syriaca* L. Botany Letters 168: 422-451.

Fremstad, E. (2010): NOBANIS – Invasive Alien Species Fact Sheet – *Lupinus polyphyllus*. http://www.nobanis.org

Fried, G., Caño, L., Brunel, S., Beteta, E., Charpentier, A., Herrera, M., Starfinger, U. & Panetta, F.D. (2016): Monographs on invasive plants in Europe: *Baccharis halimifolia* L. Botany Letters 163: 127-153.

Gaertner, M., Breeyen, A.D., Hui, C. & Richardson, D.M. (2009): Impacts of alien plant invasions on species richness in Mediterranean-type ecosystems: a meta-analysis. Progress in Physical Geography 33: 319-338.

Galle, R., Erdelyi, N., Szpisjak, N., Tolgyesi, C. & Maak, I. (2015): The effect of the invasive *Asclepias syriaca* on the ground-dwelling arthropod fauna. Biologia 70: 104-112.

Gerber, E., Krebs, C., Murrell, C., Moretti, M., Rocklin, R. & Schaffner, U. (2008): Exotic invasive knotweeds (*Fallopia* spp.) negatively affect native plant and invertebrate assemblages in European riparian habitats. Biological Conservation 141: 646-654.

Gibson, M.R., Pauw, A. & Richardson, D.M. (2013): Decreased insect visitation to a native species caused by an invasive tree in the Cape Floristic Region. Biological Conservation 157: 196-203.

Giuliano, W.M., Accamando, A.K. & McAdams, E.J. (2004): Lepidoptera-habitat relationships in urban parks. Urban Ecosystems 7: 361-370.

Grace, J. & Nelson, M. (1981): Insects and their pollen loads at a hybrid *Heracleum* site. New Phytologist 87: 413-423.

Hanslin, H.M. & Kollmann, J. (2016): Positive responses of coastal dune plants to soil conditioning by the invasive *Lupinus nootkatensis.* Acta Oecologia 77: 1-9.

Hanula, J.L. & Horn, S. (2011a): Removing an exotic shrub from riparian forests increases butterfly abundance and diversity. Forest Ecology and Management 262: 674-680.

Hanula, J.L. & Horn, S. (2011b): Removing an invasive shrub (Chinese privet) increases native bee diversity and abundance in riparian forests of the southeastern United States. Insect Conservation and Diversity 4: 275-283.

Hejda, M. & Pyšek, P. (2008): Estimating the community-level impact of the riparan alien species *Mimulus guttatus* by using a replicated BACI field experiment. In: Rabitsch, W., Essl, F. & Klingenstein, F. (eds) Biological Invasions – from Ecology to Conservation. Neobiota 7: 250-257.

Hejda, M., Pyšek, P. & Jarošík, V. (2009): Impact of invasive plants on the species richness, diversity and composition of invaded communities. Journal of Ecology 97: 393-403.

Helmisaari, H. (2010): NOBANIS - Invasive alien species fact sheet - *Impatiens glandulifera*. Online data-base of the European Network on Invasive Alien Species – NOBANIS. www.nobanis.org (online 11/2018).

Herrera, A.M. & Dudley, T.L. (2003): Reduction of riparian arthropod diversity as a consequence of giant reed (*Arundo donax*) invasion. Biological Invasions 5: 167-177.

Higgins, G.T. (2008): Irish Wildlife Manual. *Rhododendron ponticum*: A guide to management on nature conservation sites. https://www.npws.ie/sites/default/files/publications/pdf/IWM33.pdf

Hocking, S., Toop, T., Jones, D., Graham, I., & Eastwood, D. (2023): Assessing the relative impacts and economic costs of Japanese knotweed management methods. Scientific Reports 13: 1-14.

Hoffmann, J.H., Moran, V.C. & Impson, F.A.C. (1998): Promising results from the first biological control programme against a solanaceous weed (*Solanum elaeagnifolium*). Agriculture, Ecosystems & Environment 70: 145-150.

Hudson, J.R., Hanula, J.L. & Horn, S. (2013): Removing Chinese privet from riparian forests still benefits pollinators five years later. Biological Conservation 167: 355-362.

Info Flora (2022): Beurteilung der Invasivität von drei, in der Schweiz vorkommenden, gebietsfremden Arten der Gattung Conyza: *C. bonariensis, C. canadensis* und C. *sumatrensis*. Beurteilung und Bericht der Neophyten Expertengruppe von Info Flora. (Accessed 3.8.2023). https://www.infoflora.ch/assets/content/documents/neophytes/Conyza_Bericht_20220608_D_final.pdf

IPBES (2017): The assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production. Potts, S.G., Imperatriz-Fonseca, V.L. & Ngo, H.T. (eds) Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany. 552 pp.

Issaly, E.A., Sérsic, A.N., Pauw, A., Cocucci, A.A., Traveset, A., Benítez-Vieyra, S.M. & Paiaro, V. (2020): Reproductive ecology of the bird-pollinated *Nicotiana glauca* across native and introduced ranges with contrasting pollination environments. Biological Invasions 22: 485-498.

IUCN (2019a): Management measures in relation to invasive alien species threatening wild pollinators in the EU – *Rhododendron ponticum*. Technical note prepared by IUCN for the European Commission.

IUCN (2019b): Management measures in relation to invasive alien species threatening wild pollinators in the EU – *Solidago*. Technical note prepared by IUCN for the European Commission.

IUCN (2020): Managing invasive alien species to protect wild pollinators. Technical guidance prepared for the European Commission under contract No 07.0202/2018/795538/SER/ENV.D.2 "Technical support related to the implementation of the EU Pollinators Initiative". 46 pp.

Jakobsson, A. & Padrón, B. (2014): Does the invasive *Lupinus polyphyllus* increase pollinator visitation to a native herb through effects on pollinator population sizes? Oecologia 174: 217-226.

Jakobsson, A., Padrón, B. & Ågren, J. (2015): Distance-dependent effects of invasive *Lupinus polyphyllus* on pollination and reproductive success of two native herbs. Basic and applied ecology 16: 120-127.

Jakobsson, A., Padrón, B. & Traveset, A. (2008): Pollen transfer from invasive *Carpobrotus* spp. to natives – A study of pollinator behaviour and reproduction success. Biological Conservation 141: 136-145.

Jakobsson, A., Padrón, B. & Traveset, A. (2009): Competition for pollinators between invasive and native plants: Effects of spatial scale of investigation (note). Écoscience 16: 138-141.

Jiménez-Ruiz, J., Hardion, L., Del Monte, J.P., Vila, B. & Santín-Montanyá, M.I. (2021): Monographs on invasive plants in Europe N°4: *Arundo donax* L. Botany Letters 168: 131-151.

Johnson, C.A., Dutt, P. & Levine, J.M. (2022): Competition for pollinators destabilizes plant coexistence. Nature 607: 721-725.

Kapilkumar, I., Galle-Szpisjak, N., Kaur, H. & Galle, R. (2019): Forest type interacts with Milkweed invasion to affect spider communities. Insect Conservation and Diversity 12: 321-328.

Karlovarský kraj (2015): Metodiky likvidace invazních druhů rostlin. Vydání první. Karlovarský kraj.

Kasiotis, K.M., Evergetis, E., Papachristos, D., Vangelatou, O., Antonatos, S., Milonas, P., Haroutounian, S.A. & Machera, K. (2020): An essay on ecosystem availability of *Nicotiana glauca* Graham alkaloids: the honey-bees case study. BMC ecology 20: 57.

Katz, G.L. & Shafroth, P.B. (2003): Biology, ecology and management of *Elaeagnus angustifolia* L. (Russian olive) in Western North America. Wetlands 23: 763-777.

Kelemen, A., Valko, O., Kroel-Dulay, G., Deak, B., Torok, P., Toth, K., Miglecz, T. & Tothmeresz, B. (2016): The invasion of common milkweed (*Asclepias syriaca*) in sandy old-fields - is it a threat to the native flora? Applied Vegetation Science 19: 218-224.

Koopman, K.R., Beringen, R., Collas, F.P.L., Matthews, J., Odé, B., Pot, R., Sparrius, L.B., van Valkenburg, J.L.C.H., Verbrugge, L.N.H. & Leuven, R.S.E.W. (2012): Knowledge document for risk analysis of the non-native Monkeyflower (*Mimulus guttatus*) in the Netherlands. Department of Environmental Science, Faculty of Science, Institute for Water and Wetland Research, Radboud University Nijmegen. 46 pp.

Kovács-Hostyánszki, A., Szigeti, V., Miholcsa, Z., Sándor, D., Soltész, Z., Török, E. & Fenesi, A. (2022): Threats and benefits of invasive alien plant species on pollinators. Basic and Applied Ecology 64: 89-102.

Kudrnovsky, H., Ellmauer, T., Götzl, M., Paternoster, D., Sonderegger, G. & Schwaiger, E. (2020): Report for a list of Annex I habitat types important for Pollinators. ETC/BD report to the EEA.

Lapin, K. (2017): Information on measures and related costs in relation to species included on the Union list: *Asclepias syriaca*. Technical note prepared by IUCN for the European Commission. https://circabc.europa.eu/ui/group/4cd6cb36-b0f1-4db4-915e-65cd29067f49/library/854cb8c6-fd41-4ce6-92ed-8c431fe1b959/details

Lazzaro, L., Ferretti, G., Bianchi, E. & Benesperi, R. (2019): Treatment by glyphosate-based herbicide allowed recovering native species after *Oxalis pes-caprae* L. invasion: indications from a Mediterranean island. Plant Biosystems 153: 651-659.

Lechner, Y., Maschek, O., Kirisits, T. & Halmschlager, E. (2023): Further pathogenicity testing of *Verticillium nonalfalfae*, a biocontrol agent against the invasive Tree of Heaven (*Ailanthus altissima*), on non-target tree species in Europe. Phytoparasitica 51: 113-130.

Lehner, S., Schulz, S. & Dötterl, S. (2022): The mystery of the butterfly bush *Buddleja davidii*: How are the butterflies attracted? Frontiers in Plant Science 13: 994851.

León, Y.M., Carreira, C.R., Jiménez, R.E., Almunia, J., Wiemers, M., Vujic, A., Macadam, C., Raser, J. & Hochkirch, A. (2023): Canarian Islands endemic pollinators of the Laurel Forest zone - Conservation plan 2023-2028. Publication prepared for the European Commission within the framework of the contract No 07.0202/2020/839411/SER/ENV.0.2

Lesieur, V., Sforza, R.F.H., Sheppard, A.W. & Shaw, R.H. (2023): Prioritising environmental invasive weeds of European concern for classical biological control: A reanalysis. Weed Research 63: 218-231.

Liao, C., Peng, R., Luo, Y., Zhou, X., Wu, X., Fang, C., Chen, J. & Li, B. (2007): Altered ecosystem carbon and nitrogen cycles by plant invasion: a meta-analysis. New Phytologist 177: 706-714.

Litt, A.R., Cord, E.E., Fulbright, T.E. & Schuster, G.L. (2014): Effects of invasive plants on arthropods. Conservation Biology 28: 1532-1549.

Lopezaraiza-Mikel, M.E., Hayes, R.B., Whalley, M.R. & Memmott, J. (2007): The impact of an alien plant on a native plant-pollinator network: An experimental approach. Ecology Letters 10: 539-550.

Loydi, A., Donath, T.W., Eckstein, R.L. & Otte, A. (2015): Non-native species litter reduces germination and growth of resident forbs and grasses: allelopathic, osmotic or mechanical effects? Bioloical Invasions 17: 581-595.

Lozan, A.I., Monaghan, M.T., Spitzer, K., Jaroš, J., Žurovcová, M. & Brozcaron, V. (2008): DNA-based confirmation that the parasitic wasp *Cotesia glomerata* (Braconidae, Hymenoptera) is a new threat to endemic butterflies of the Canary Islands. Conservation Genetics 9: 1431-1437.

Maceda-Veiga, A., Basas, E., Lanzaco, G., Sala, M., de Sostoa, A. & Serra, A. (2016): Impacts of the invader giant reed (*Arundo donax*) on riparian habitats and ground arthropod communities. Biological Invasions 18: 731-749.

Maerz, J.C., Blossey, B. & Nuzzo, V. (2005): Green frogs show reduced foraging success in habitats invaded by Japanese knotweed. Biodiversity and Conservation 14: 2901-2911.

Mallard, F. (2008): Effet d'une espèce végétale introduite envahissante le Séneçon en arbre *Baccharis halimifolia* sur le peuplement d'arthropodes dans le Golfe du Morbihan. MSc dissertation, Université de Rennes.

Marchante, H., López-Núñez, F.A., Freitas, H., Hoffmann, J.H., Impson, F. & Marchante, E. (2017): First report of the establishment of the biocontrol agent *Trichilogaster acaciaelongifoliae* for control of invasive *Acacia longifolia* in Portugal. EPPO Bulletin 47: 274-278.

Marshall, G. (1987): A review of the biology and control of selected weed species in the genus *Oxalis: O. stricta* L., *O. latifolia* H.B.K. and *O. pes-caprae*. Crop Protection 6: 355-364.

Maschek, O. & Halmschlager, E. (2018): Effects of *Verticillium nonalfalfae* on *Ailanthus altissima* and associated indigenous and invasive tree species in eastern Austria. European Journal of Forest Research 137: 197-209.

McKinney, A.M. & Goodell, K. (2010): Shading by invasive shrub reduces seed production and pollinator services in a native herb. Biological Invasions 12: 2751-2763.

Mihulka, S., Pyšek, P., Martínková, J. & Jarošík, V. (2006): Invasiveness of *Oenothera congeners* alien to Europe: Jack of all trades, master of invasion? Perspectives in Plant Ecology, Evolution and Systematics 8: 83-96.

Moragues, E. & Traveset, A. (2005): Effect of *Carpobrotus* spp. on the pollination success of native plant species of the Balearic Islands. Biological Conservation 122: 611-619.

Morales, C.L. & Traveset, A. (2009): A meta-analysis of impacts of alien vs. native plants on pollinator visitation and reproductive success of co-flowering native plants. Ecology Letters 12: 716-728.

Moroń, D., Lenda, M., Skórka, P., Szentgyörgyi, H., Settele, J. & Woyciechowski, M. (2009): Wild pollinator communities are negatively affected by invasion of alien goldenrods in grassland landscapes. Biological Conservation 142: 1322-1332.

Najberek, K., Solarz, W., Wysoczański, W., Węgrzyn, E. & Olejniczak, P. (2023): Flowers of *Impatiens glandulifera* as hubs for both pollinators and pathogens. NeoBiota 87: 1-26.

Nielsen, C., Heimes, C. & Kollmann, J. (2008): Little evidence for negative effects of an invasive alien plant on pollinator services. Biological Invasions 10: 1353-1363.

Nielsen, C., Ravn, H.P., Cock, M. & Nentwig, W. (eds) (2005): The giant hogweed best practice manual. Guidelines for the management and control of an invasive alien weed in Europe. Forest and Landscape Denmark, Hoersholm, Denmark.

Nienhuis, C.M., Dietzsch, A.C. & Stout, J.C. (2009): The impacts of an invasive alien plant and its removal on native bees. Apidologie 40: 450-463.

Nieto, A., Roberts, S.P.M., Kemp, J., Rasmont, P., Kuhlmann, M., García Criado, M., Biesmeijer, J.C., Bogusch, P., Dathe, H.H., De la Rúa, P., De Meulemeester, T., Dehon, M., Dewulf, A., Ortiz-Sánchez, F.J., Lhomme, P., Pauly, A., Potts, S.G., Praz, C., Quaranta, M., Radchenko, V.G., Scheuchl, E., Smit, J., Straka, J., Terzo, M., Tomozii, B., Window, J. & Michez, D. (2014): European Red List of bees. Luxembourg: Publication Office of the European Union. 98 pp.

Novoa, A., Brundu, G., Day, M. D., Deltoro, V., Essl, F., Foxcroft, L. C., ... & Wilson, J. R. (2019): Global actions for managing cactus invasions. Plants 8: 421.

Novoa, A., Hirsch, H., Castillo, M.L., Canavan, S., González, L., Richardson, D.M., Pyšek, P., Rodríguez, J., Borges Silva, L., Brundu, G., D'Antonio, C.M., Gutiérrez, J.L., Mathese, M., Levin, S., Silva, L. & Le Roux, J.J. (2023): Genetic and morphological insights into the *Carpobrotus* hybrid complex around the world. NeoBiota 89: 135-160.

Oliver, B.W., Berge, T.W., Solhaug, K.A. & Fløistad, I.S. (2020): Hot water and cutting for control of *Impatiens glandulifera*. Invasive Plant Science and Management 13: 84-93.

Ollerton, J., Watts, S., Connerty, S., Lock, J., Parker, L., Wilson, I., Schueller, S., Nattero, J., Cocucci, A. A., Izhaki, I., Geerts, S., Pauw, A. & Stout, J.C. (2012): Pollination ecology of the invasive tree tobacco *Nicotiana glauca*: comparisons across native and non-native ranges. Journal of Pollination Ecology 9: 85-95.

Pace, M.C. & Cameron, K.M. (2017): The Systematics of the *Spiranthes cernua* species complex (Orchidaceae): untangling the Gordian knot. Systematic Botany 42: 640-669.

Padrón, B., Traveset, A., Biedenweg, T., Diaz, D., Nogales, M. & Olesen, J.M. (2009): Impact of alien plant invaders on pollination networks in two archipelagos. PLoS One 4: e6275.

Parra-Tabla, V. & Arceo-Gómez, G. (2021): Impacts of plant invasions in native plant-pollinator networks. New Phytologist 230: 2117-2128.

Pendleton, R.L., Pendleton, B.K. & Finch, D. (2011): Displacement of native riparian shrubs by woody exotics: effects on arthropod and pollinator community composition. Natural Resources and Environmental Issues 16: 25.

Pergl, J. (2017): Information on measures and related costs in relation to species included on the Union list: *Heracleum mantegazzianum*. Technical note prepared by IUCN for the European Commission.

Pergl, J. (2019): Information on measures and related costs in relation to species included on the Union list: *Heracleum mantegazzianum*, *Heracleum sosnowskyi*, *Heracleum persicum*. Technical note prepared by IUCN for the European Commission. https://circabc.europa.eu/ui/group/4cd6cb36-b0f1-4db4-915e-65cd29067f49/library/854cb8c6-fd41-4ce6-92ed-8c431fe1b959/details

Pergl, J., Härtel, H., Pyšek, P. & Stejskal, R. (2020): Don't throw the baby out with the bathwater – ban of glyphosate use depends on context. NeoBiota 56: 27-29.

Poblador, S., Lupon, A., Martí, E., Sabater, F., Sabaté, S. & Bernal, S. (2019): The influence of the invasive alien nitrogen-fixing *Robinia pseudoacacia* L. on soil nitrogen availability in a mixed Mediterranean riparian forest. European Journal of Forest Research 138: 1083-1093.

Pollard, K.M., Varia, S., Seier, M.K. & Ellison, C.A. (2021): Battling the biotypes of balsam: the biological control of *Impatiens glandulifera* using the rust fungus *Puccinia komarovii* var. *glanduliferae* in GB. Fungal Biology 125: 637-645.

Potter, S. & Sheehan, M.R. (2017): Opuntioid Cacti Management Guide: Best practice information for *Austrocylindropuntia*, *Cylindropuntia* and *Opuntia* species in Australia. Department of Primary Industries and Regional Development (WA), Perth.

Proesmans, W., Albrecht, M., Gajda, A., Neumann, P., Paxton, R.J., Pioz, M., Polzin, C., Schweiger, O., Settele, J., Szentgyörgyi, H., Thulke, H.-H. & Vanbergen, A.J. (2021): Pathways for novel epidemiology: plant–pollinator–pathogen networks and global change. Trends in ecology & evolution 36: 623-636.

Prowse, A. & Goodridge, F. (2000): Pollinator visitation rates of *Impatiens glandulifera* and other native riparian vegetation. Aspects of Applied Biology 58: 249-254.

Pyšek, P., Cock, M.J.W., Nentwig, W. & Ravn, H.P. (2007): Ecology and Management of Giant Hogweed (*Heracleum mantegazzianum*). CAB International. 324 pp.

Pyšek, P., Jarošík, V., Hulme, P.E., Pergl, J., Hejda, M., Schaffner, U. & Vilà, M. (2012): A global assessment of invasive plant impacts on resident species, communities and ecosystems: the interaction of impact measures, invading species' traits and environment. Global Change Biology 18: 1725-1737.

Rabitsch, W., Kudrnovsky, H. & Götzl, M. (2021): Invasive alien plant species, habitat types important for pollinators, and the possible risks in the European Union. ETC/BD report to the EEA. 56 pp.

Rajmis, S., Thiele, J. & Marggraf, R. (2016): A cost-benefit analysis of controlling giant hogweed (*Heracleum mantegazzianum*) in Germany using a choice experiment approach. NeoBiota 31: 19-41.

Ramula, S. (2020): Annual mowing has the potential to reduce the invasion of herbaceous *Lupinus polyphyllus*. Biological Invasions 22: 3163-3173.

Reif, J., Hanzelka, J., Kadlec, T., Strobl, M. & Hejda, M. (2016): Conservation implications of cascading effects among groups of organisms: The alien tree *Robinia pseudacacia* in the Czech Republic as a case study. Biological Conservation 198: 50-59.

Renner, S.S., Graf, M.S., Hentschel, Z., Krause, H. & Fleischmann, A. (2021): High honeybee abundances reduce wild bee abundances on flowers in the city of Munich. Oecologia 195: 825-831.

Rivest, S. & Forrest, J.R.K. (2019): Defence compounds in pollen: why do they occur and how do they affect the ecology and evolution of bees? New Phytologist 225: 1053-1064.

Roberts, J. & Florentine, S. (2022): Biology, distribution and management of the globally invasive weed *Solanum elaeagnifolium* Cav (silverleaf nightshade): A global review of current and future management challenges. Weed Research 62: 393-403.

Robertson, M.P., Harris, K.R., Coetzee, J.A., Foxcroft, L.C., Dippenaar-Schoeman, A.S. & van Rensburg, B.J. (2011): Assessing local scale impacts of *Opuntia stricta* (Cactaceae) invasion on beetle and spider diversity in Kruger National Park, South Africa. African Zoology 46: 205-223.

Rodríguez-Caballero, G., Roldán, A. & Caravaca, F. (2020): Invasive *Nicotiana glauca* shifts the soil microbial community composition and functioning of harsh and disturbed semiarid Mediterranean environments. Biological Invasions 22: 2923-2940.

Ross, C.A. & Auge, H. (2008): Invasive *Mahonia* plants outgrow their native relatives. Plant Ecology 199: 21-31.

Sádlo, J., Vítková, M., Pergl, J. & Pyšek, P. (2017): Towards site-specific management of invasive alien trees based on the assessment of their impacts: the case of *Robinia pseudoacacia*. NeoBiota 35: 1-34.

Saegesser, J., Fischer, D. & Fischer, K. (2016): Long-term control of *Impatiens glandulifera* in a Swiss Forest: does zero-tolerance pay off? In: Ries, C., Krippel, Y. (eds), Biological invasions: interactions with environmental change. Book of abstracts. NEOBIOTA 2016 - 9th International Conference on Biological Invasions, p. 69. Fondation fauneflore, Vianden.

Scheper, J., Reemer, M., van Kats, R., Ozinga, W.A., van der Linden, G.T.J., Schaminée, J.H.J., Siepel, H. & Kleijn, D. (2014): Museum specimens reveal loss of pollen host plants as key factor driving wild bee decline in The Netherlands. Proceedings of the National Academy of Sciences 111: 17552-17557.

Schnitzler, A. & Essl, F. (2015): From horticulture and biofuel to invasion: the spread of *Miscanthus* taxa in the USA and Europe. Weed Research 55: 221-225.

Schweiger, O., Biesmeijer, J., Bommarco, B., Hickler, T., Hulme, P.E., Klotz , S., Kühn, I., Moora, M., Nielsen, A., Ohlemüller, R., Petanidou, T., Potts, S.G., Pyšek, P., Stout, J.C., Sykes, M.T., Tscheulin, T., Vilà, M., Walther, G.-R., Westphal, C., Winter, M., Zobel, M. & Settele, J. (2010): Multiple stressors on biotic interactions: how climate change and alien species interact to affect pollination. Biological Reviews 85: 777-795.

Shaw, R.H., Bryner, S. & Tanner, R. (2009): The life history and host range of the Japanese knotweed psyllid, *Aphalara itadori* Shinji: Potentially the first classical biological weed control agent for the European Union. Biological Control 49: 105-113.

Sheppard, A.W., Paynter, Q., Mason, P., Murphy, S., Stoett, P., Cowan, P., Brodeur, J., Warner, K., Villegas, C., Shaw, R., Hinz, H., Hill, M. & Genovesi, P. (2019): IUCN SSC Invasive Species Specialist Group. The Application of Biological Control for the Management of Established Invasive Alien Species Causing Environmental Impacts. https://www.cbd.int/doc/publications/cbd-ts-91-en.pdf

Somogyi, A.A., Lőrinczi, G., Kovacs, J. & Maak, I. (2017): Structure of ant assemblages in planted poplar (*Populus alba*) forests and the effect of the common milkweed (*Asclepias syriaca*). Acta Zoologica Academiae Scientiarum Hungaricae 63: 443-457.

Stephenson, A.G. (1981): Toxic nectar deters nectar thieves of *Catalpa speciosa*. American Midland Naturalist 105: 381-383.

Stephenson, A.G. (1982): Iridoid glycosides in the nectar of *Catalpa speciosa* are unpalatable to nectar thieves. Journal of Chemical Ecology 8: 1025-1034.

Stevenson, P.C., Nicolson, S.W. & Wright, G.A. (2017): Plant secondary metabolites in nectar: impacts on pollinators and ecological functions. Functional Ecology 31: 65-75.

Stout, J.C. & Casey, L.M. (2014): Relative abundance of an invasive alien plant affects insect–flower interaction networks in Ireland. Acta Oecologica 55: 78-85.

Stout, J.C. & Morales, C.L. (2009): Ecological impacts of invasive alien species on bees. Apidologie 40: 388-409.

Stout, J.C. & Tiedeken, E.J. (2016): Direct interactions between invasive plants and native pollinators: evidence, impacts and approaches. Functional Ecology 31: 38-46.

Stout, J.C. (2007): Pollination of invasive *Rhododendron ponticum* (Ericaceae) in Ireland. Apidologie 38: 198-206.

Strgulc Krajšek, S., Bahčič, E., Čoko, U. & Dolenc Koce, J. (2020): Disposal methods for selected invasive plant species used as ornamental garden plants. Management of Biological Invasions 11: 293-305.

Suadicani, K., Buttenschøn, R.M., Ravn, H.P. & Johannsen, V.K. (2017): Kæmpe-bjørneklo i Danmark – status for bekæmpelsen. Udbredelse og indsats, de anvendte metoder og deres effekt samt en analyse af samfundsøkonomien i forbindelse med bekæmpelsen. IGN, Frederiksberg. 55 pp.

Sudnik-Wójcikowska, B., Moysiyenko, I., Slim, P.A. & Moraczewski, I.R. (2009): Impact of the invasive species *Elaeagnus angustifolia* L. on vegetation in pontic desert steppe zone (Southern Ukraine). Polish Journal of Ecology 57: 269-281.

Suehs, C.M., Affre, L. & Medail, F. (2005): Unexpected insularity effects in invasive plant mating systems: the case of *Carpobrotus* (Aizoaceae) taxa in the Mediterranean Basin. Biological Journal of the Linnean Society 85: 65-79.

Szigeti, V., Fenesi, A., Soltesz, Z., Berki, B. & Kovacs-Hostyanszki, A. (2020): Neutral effect of an invasive plant species with specialized flower structure on native pollinator communities. Biological Invasions 22: 3017-3030.

Tallent-Halsell, N.G. & Watt, M.S. (2009): The invasive *Buddleja davidii* (Butterfly Bush). The Botanical Review 75: 292-325.

Tanner, R. (2017): Information on measures and related costs in relation to species included on the Union list: *Impatiens glandulifera*. Technical note prepared by IUCN for the European Commission.

Tataridas, A., Moreira, M., Frazão, L., Kanatas, P., Ota, N. & Travlos, I. (2023): Biology of Invasive Plants 5. *Solanum elaeagnifolium* Cav. Invasive Plant Science and Management 16: 139-159.

Tenorio-Escandón, P., Ramírez-Hernández, A., Flores, J., Juan-Vicedo, J. & Martínez-Falcón, A.P. (2022): A systematic review on *Opuntia* (Cactaceae; Opuntioideae) flower-visiting insects in the world with emphasis on Mexico: implications for biodiversity conservation. Plants 11: 131.

Thijs, K.W., Brys, R., Verboven, H.A.F. & Hermy, M. (2012): The influence of an invasive plant species on the pollination success and reproductive output of three riparian plant species. Biological Invasions 14: 355-365.

Tiedeken, E.J. & Stout, J.C. (2015): Insect-flower interaction network structure is resilient to a temporary pulse of floral resources from invasive *Rhododendron ponticum*. PLoS One 10: e0119733.

Tiedeken, E.J., Egan, P.A., Stevenson, P.C., Wright, G.A., Brown, M.J.F., Power, E.F., Farrell, I., Matthews, S.M. & Stout, J.C. (2016): Nectar chemistry modulates the impact of an invasive plant on native pollinators. Functional Ecology 30: 885-893.

Tiedeken, E.J., Stout, J.C., Egan, P., Stevenson, P.C. & Wright, G.A. (2014): Bumblebees are not deterred by ecologically relevant concentrations of nectar toxins. Journal of Experimental Biology 217: 1620-1635.

Tiley, G.E.D., Dodd, F.S. & Wade, P.M. (1996): *Heracleum mantegazzianum* Sommier & Levier. Journal of Ecology 84: 297-319.

Traveset, A. & Richardson, D.M. (2006): Biological invasions as disruptors of plant-animal reproductive mutualisms. Trends in Ecology and Evolution 21: 208-216.

Truscott, A.M., Palmer, S.C.F., Soulsby, C. & Hulme, P.E. (2008a): Assessing the vulnerability of riparian vegetation to invasion by *Mimulus guttatus*: relative importance of biotic and abiotic variables in determining species occurrence and abundance. Diversity & Distributions 14: 412-421.

Truscott, A.M., Palmer, S.C.F., Soulsby, C., Westaway, S. & Hulme, P.E. (2008b): Consequences of invasion by the alien plant *Mimulus guttatus* on the species composition and soil properties of riparian plant communities in Scotland. Perspectives in Plant Ecology, Evolution and Systematics 10: 231-240.

Tscheulin, T. & Petanidou, T. (2013): The presence of the invasive plant *Solanum elaeagnifolium* deters honeybees and increases pollen limitation in the native co-flowering species *Glaucium flavum*. Biological Invasions 15: 385-393.

Tscheulin, T., Petanidou, T., Potts, S.G. & Settele, J. (2009): The impact of *Solanum elaeagnifolium*, an invasive plant in the Mediterranean, on the flower visitation and seed set of the native co-flowering species *Glaucium flavum*. Plant Ecology 205: 77-85.

Valtonen, A., Jantunen, J. & Saarinen, K. (2006): Flora and lepidoptera fauna adversely affected by invasive *Lupinus polyphyllus* along road verges. Biological Conservation 133: 389-396.

Van Swaay, C., Cuttelod, A., Collins, S., Maes, D., López Munguira, M., Šašić, M., Settele, J., Verovnik, R., Verstrael, T., Warren, M., Wiemers, M. & Wynhof, I. (2010): European Red List of Butterflies Luxembourg: Publications Office of the European Union. 61 pp.

Vanbergen, A.J., Espíndola, A. & Aizen, M.A. (2018): Risks to pollinators and pollination from invasive alien species. Nature ecology & evolution 2: 16-25.

Vanparys, V., Cawoy, V., Mahaux, O. & Jacquemart, A.L. (2011): Comparative study of the reproductive ecology of two co-occurring related plant species: the invasive *Senecio inaequidens* and the native *Jacobaea vulgaris*. Plant Ecology and Evolution 144: 3-11.

Vanparys, V., Meerts, P. & Jacquemart, A.-L. (2008): Plant-pollinator interactions: comparison between an invasive and a native congeneric species. Acta Oecologia 34: 361-369.

Verloove, F., Zonneveld, B.J.M. & Semple, J.C. (2017): First evidence for the presence of invasive *Solidago altissima* (Asteraceae) in Europe. Willdenowia 47: 69-75.

Vilà, M., Bartomeus, I., Dietzsch, A.C., Petanidou, T., Steffan-Dewenter, I., Stout, J.C. & Tscheulin, T. (2009): Invasive plant integration into native plant–pollinator networks across Europe. Proceedings of the Royal Society B: Biological Sciences 276: 3887-3893.

Vilà, M., Espinar, J.L., Hejda, M., Hulme, P.E., Jarošík, V., Maron, J.L., Pergl, J., Schaffner, U., Sun, Y. & Pyšek, P. (2011): Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities and ecosystems. Ecology Letters 14: 702-708.

Vujić, A., Gilbert, F., Flinn, G., Englefield, E., Ferreira, C.C., Varga, Z., Eggert, F., Woolcock, S., Böhm, M., Mergy, R., Ssymank, A., van Steenis, W., Aracil, A., Földesi, R., Grković, A., Mazanek, L, Nedeljković, Z., Pennards, G.W.A., Pérez, C., Radenković, S., Ricarte, A., Rojo, S., Ståhls, G., van der Ent, L.-J., van Steenis, J., Barkalov, A., Campoy, A., Janković, M., Likov, L., Lillo, I., Mengual, X., Milić, D., Miličić, M., Nielsen, T., Popov, G., Romig, T., Šebić, A., Speight, M., Tot, T., van Eck, A., Veselić, S., Andric, A., Bowles, P., De Groot, M., Marcos-García, M.A., Hadrava, J., Lair, X. , Malidžan, S., Nève, G., Obreht Vidakovic, D., Popov, S., Smit, J.T., Van De Meutter, F., Veličković, N. & Vrba, J. (2022): Pollinators on the edge: our European hoverflies. The European Red List of Hoverflies. Brussels, Belgium: European Commission. 108 pp.

Wiemers, M., Ellis, S., Cosma, E., Sevilleja, C., van Swaay, C.A.M., Wynhoff, I., Gallego-Zamorano, J. & Teixeira, S.B.M. (2022): Species Action Plan for the Madeiran Large White *Pieris wollastoni*. Butterfly Conservation Europe. 19 pp.

Willow, J., Tamayo, M. & Jóhannsson, M.H. (2017): Potential impact of Nootka lupine (*Lupinus nootkatensis*) invasion on pollinator communities in Iceland. Icelandic Agricultural Sciences 30: 51-54.

Wu, H. (2007): The biology of Australian weeds 49. *Conyza bonariensis* (L.) Cronquist. Plant Protection Quarterly 22: 122-131.

Wu, H., Stanton, R. & Lemerle, D. (2016): Herbicidal control of *Solanum elaeagnifolium* Cav. in Australia. Crop Protection 88: 58-64.

Zaninotto, V., Thebault, E. & Dajoz, I. (2023): Native and exotic plants play different roles in urban pollination networks across seasons. Oecologia 201: 525-536.

Zlatkov, B., Beshkov, S. & Ganeva, T. (2018): *Oenothera speciosa* versus *Macroglossum stellatarum*: killing beauty. Arthropod-Plant Interactions 12: 395-400.

Zumkier, U. (2012): Impacts of the invasive alien *Heracleum mantegazzianum* on native plant-pollinator interactions. Thesis Univ. Bielefeld. 116 pp.