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“Whatever makes the past, the distant, or the future, predominate over the present, advances us in the dignity of thinking beings.”

Samuel Johnson (1791)

“In the 1950s, the planet still had isolated islands, in both geographical and cultural terms - lands of unique mysteries, societies, and resources. By the end of the 20th century, expanding numbers of people, powerful technology, and economic demands had linked Earth’s formerly isolated, relatively non-industrialized places with highly developed ones into an expansive and complex network of ideas, materials, and wealth.”

Lutz Warren and Kieffer (2010)

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FOREWORD

This Ph.D. thesis is the result of my time as project employee at the Division of Conservation Biology, Vegetation Ecology & Landscape Ecology, Department of Botany and Biodiversity Research, Faculty of Life Sciences at the University of Vienna, Austria during the years 2014-2017. The project was funded by the ERA-Net BiodivERsA, with the FWF (Austrian Science Fund, project “Who Is Next” I-1443-B25) as national funder.

The thesis introduces into the field of invasion biology, as all my publications deal with aspects of it. After shortly explaining different statistical models, as essential parts of my studies, I formulate the research questions. Subsequently, I present four manuscripts. Three of them have already been published in peer-reviewed journals. The last one has been submitted and is currently under consideration.

ABSTRACT

Within the last two centuries the spread of invasive species, i.e. human-introduced non-native species that manage to establish self-sustaining populations and spread into multiple sites, has caused important problems to biodiversity, economy and human well-being. Consequently, identifying drivers of invasions, future areas of risk as well as potential future invaders has become a scientific focus and national and international regulatory frameworks for alien species management have been developed.

Within my dissertation, I first analysed plant species’ trait profiles and their power for explaining invasiveness by comparing species recorded and not recorded as invasive aliens anywhere in the world, respectively. My results suggested moderate predictive ability of species traits. However, multidimensional variability in trait space was larger among invasive than among non-invasive species suggesting that invasion success has a considerable idiosyncratic component and is probably highly context specific.

Using different modelling approaches I assessed changes in invasion risk from garden plants in Europe under a warming climate, with garden plants being one of the most important source pools for potential invaders. Additionally, I evaluated whether hybridization between

garden and native plant species will increase, and disentangled the relative effects of climate warming and propagule pressure on invasion success. While climate warming does not seem to increase hybridization risk, hotspots of naturalization risk will increase considerably. Using niche-demographic models I found that future areas simulated to be occupied by invading garden species are not larger but smaller than under a constant climate. The likely reason is a spatial displacement of suitable sites over the course of the century that drives transiently established populations to extinction and creates an invasion debt. In addition, I found restrictions of use to have a strong, non-linear effect on species spread, implying that important benefits only accrue if restrictions successfully reduce cultivation intensity to low levels.

INTRODUCTION

Within the last century the spread of non-native species has resulted in harm to biodiversity, economy and human well-being (Vilà *et al.*, 2011; Simberloff *et al.*, 2013; Blackburn *et al.*, 2014). The tendency of some non-native species to successfully spread and display invasive behaviour did already intrigue famous scientists like Charles Darwin, Alphonse de Candolle, Joseph Hooker or Joseph Grinnell (Richardson, 2008). Meanwhile there's an endless list of animals, plants, bacteria and fungi that managed to colonize new habitats around the globe at an alarming rate (Seebens *et al.*, 2017). As an example, human activity resulted in 13,168 plant species (equals the size of the European flora or 3.9% of the global vascular flora) becoming naturalized somewhere on the globe (van Kleunen *et al.*, 2015). Human intervention, both accidental and intentional, has hence led to a global environmental problem that is now one of the most important causes of extinctions of native animals and plants (Bellard *et al.*, 2016). Apart from ecological consequences, invasive species have had severe economic effects, causing damage worth billions of Euro each year.

Invasion Ecology

The British ecologist Charles Sutherland Elton (1900 – 1991), as one of the most important ecologists of his time, published a seminal work which is widely respected as the cornerstone of invasion biology (Richardson & Pysek, 2008): The ecology of invasions by animals and plants (Elton, 1958). The book demonstrates the global scale and implications of biological invasions for life on earth – or 'ecological explosion', as Elton called it - and thus provides a map for new research directions (Richardson & Pysek, 2008). Since then, the field of invasion biology has been dedicated to detecting, understanding and mitigating invasion impact. This research interest was largely motivated by the negative impact of invasive species on the conservation of biodiversity and by their increasing number, which is a foremost result of ongoing globalization (Levine & D'Antonio, 2003; van Kleunen *et al.*, 2015; Seebens *et al.*, 2017), land-use change (Chytrý *et al.*, 2012) and climate change (Bellard *et al.*, 2013). Understanding macro-ecological patterns of biological invasions (van Kleunen *et al.*, 2015), national and international regulatory frameworks for the protection of economy and human health (Hulme *et al.*, 2008; Hulme, 2012), as well as technological improvements to deal with impacts (Pyšek *et al.*, 2012; Simberloff *et al.*, 2013) are main research goals for invasion

scientists. On the following pages I want to discuss important facts about biological invasions, ranging from history, drivers and invasion hypotheses and to management instruments in science.

History and Nature of Biological Invasions

Animals or plants were already traded by medieval kingdoms before 1500 AD (Hulme, 2009). However, the exchange of non-native species gained momentum when European nations started to explore the globe. Christopher Columbus' travel to North America is a symbolic event in the early phase of this era which saw the development of a truly global network of species exchange. The year 1492 has hence been used as a threshold in invasion biology which defines all non-native species naturalized subsequently as 'neobiota' (as opposed to 'archeobiota'). European colonialism and the broad commercialization of sea trade during the early modern age fostered the transport of species between European 'home countries' and colonies but also among colonies. Part of this species exchange was intentional. In particular, economically useful plants and animals, but also pet animals and ornamental plants, were explored, cultivated and subsequently distributed across colonial empires. Many other species have, however, been transported as stowaways. Their establishment and further spread in introduced ranges was greatly facilitated by the construction of canals, highways, railways and demographic changes of human population (Hulme, 2009). This led to new interactions with native biota and in some cases to the loss of latter. At the beginning of the 19th century the rise of the Industrial Revolution led to a further strong increase in the rate of alien introductions, especially in Europe and North America (Hulme, 2009; Seebens *et al.*, 2017). Also, horticulture, as one of the most important pathways for the introduction of alien plants (Reichard & White, 2001; Dehnen-Schmutz *et al.*, 2007) and botanical gardens became increasingly popular. With the onset of globalisation after World War II trade volumes again increased and so did the exchange of non-native species (Seebens *et al.*, 2015; van Kleunen *et al.*, 2015). Recent development of new distribution channels, like e-commerce, provide additional pathways for the cross-continental flow of non-native species (Dehnen-Schmutz *et al.*, 2010; Humair *et al.*, 2015).

In order to understand the role of species and ecosystem traits in biological invasions and to be able to develop precautionary management methods or legislation instruments, one needs to understand the means of initial introduction (Hulme, 2006). Thus different attempts have

been made to classify existing pathways of introduction (e.g. 30 different pathways recorded in the Global Invasive Species Database (www.issg.org/database) or two different pathways in the Convention on Biological Diversity). One of the most recent classification schemes distinguishes six different pathways which reflect a gradient of human involvement (Hulme *et al.*, 2008): escape (intentional introduction, unintentional escape), release (intentional), contaminant (unintentional), stowaway (unintentional through a transport vector), corridor (unintentional, through infrastructures linking previously unconnected regions) and unaided (unintentional, natural dispersal of alien species across political borders). However, most of these pathways are difficult to tackle by both, management and legislation (Hulme *et al.*, 2017). In addition there's evidence for an idiosyncratic nature of biological invasions, often reflecting e.g. economical, geographical and historical attributes of a region as much as types of introduction pathways or species traits themselves (Hulme *et al.*, 2008).

Biological invasions can be categorized in a series of consecutive stages –introduction into a new region, naturalization or establishment of self-sustaining populations, and spread across suitable habitats within the introduced region (Blackburn *et al.*, 2011). The term ‘invasive’ describes species that have rapidly spread across a large area, or are about to do so. By overcoming ecological and abiotic barriers, species can pass from one stage to the next. The percentage of introduced species that manages this passage decreases with each consecutive stage (Williamson & Fitter, 1996). In addition, successful establishment of alien species introduced in the past can be delayed, reflecting past socioeconomic activities and creating an invasion debt (Essl *et al.*, 2011).

Drivers of Plant Invasions

Different drivers have been found to be responsible for the success of biological invasions, with strong variation across biogeographical regions (Taylor *et al.*, 2016). Below I emphasize a few of the most important ones, but I note that there are many additional ones, both biotic (such as biotic resistance, plant cover, vegetation type, herbivory, presence of mutualists or pathogens, seed predators) and abiotic (e.g. soil moisture, disturbance regime). This is especially true for later stages of the invasion process (Dietz & Edwards, 2006; Taylor *et al.*, 2016).

The supply of propagules, generally called propagule pressure, is widely recognized as a major determinant of invasion success (Lockwood *et al.*, 2005; Simberloff, 2009). Synonymously called introduction effort, it consists of (1) propagule size – number of individuals involved in a single release event; (2) frequency – number of release events per time unit (Lockwood *et al.*, 2005; Simberloff, 2009).

Globalisation in general is one of the most important drivers of an ever increasing propagule pressure (Levine & D'Antonio, 2003; Hulme, 2009; van Kleunen *et al.*, 2015). A general implication of spatially unlimited trade networks is a complete breakdown of dispersal limitation for species exchanged, intentionally or unintentionally, along these networks. As a corollary, historically developed biogeographical barriers become increasingly eroded (Capinha *et al.*, 2015). Apart from the exchange of goods, an increase of tourism (i.e. international passenger transport) opens new routes for propagules to remote destinations around the world (e.g. stowaways or contaminants associated with baggage, clothes, shoes; Hulme, 2009), threatening the often highly sensitive ecosystems of these isolated parts of the world. Also, as species are transported globally with increasing intensity, barriers to gene flow between once geographically separated species are reduced and risk for outbreeding depression (Bleeker *et al.*, 2007), gene swamping (Todesco *et al.*, 2016) and pollen competition (Arceo-Gómez & Ashman, 2016) might increase.

Environmental changes have contributed to plant invasions, as for example habitat degradation due to land-use change (Sala *et al.*, 2000; Chytrý *et al.*, 2008b). In addition, climate change has fostered the expansion of alien plant species to previously unsuitable regions (Walther *et al.*, 2009) and is expected to increase the number of future plant invasions (Bellard *et al.*, 2013; Early *et al.*, 2016). One of the main reasons is the disequilibrium with climatic conditions which lead to an anticipated destabilisation of resident native plant communities and decrease their biotic resistance (Svenning & Sandel, 2013). Also, climate change may increase the area climatically suitable to alien plant species (i.e. species cultivated for ornamental purpose) cultivated beyond their climatic suitable range, which may give them a head-start (Niinemets & Penuelas, 2008; Van der Veken *et al.*, 2008; Bradley *et al.*, 2012). It is expected that the combination of climate and land-use change will also increase future invasion risk of alien species in former less affected ecosystems at higher elevations (Pauchard *et al.*, 2016; Petitpierre *et al.*, 2016). Extinction events, however, will be hard to

detect since especially long – lived species will remain in their warmed habitat producing offspring that is increasingly maladapted to the changing climatic conditions, thus leading to an extinction debt (Cotto *et al.*, 2017).

The cause of invasion includes biological traits of the invading species that facilitate invasions, for example early reproduction, rapid growth rate or long-distance dispersal (Küster *et al.*, 2008; van Kleunen *et al.*, 2010; Pyšek *et al.*, 2015). Combining the search for ‘invasion’ traits with an analysis of the similarity of invasive and non-invasive species in respect to their traits could reveal whether distinct trait profiles exist that explain successful invasiveness. Apart from these factors, interspecific hybridization has been widely assumed to mitigate invasions (Ellstrand & Schierenbeck, 2000; Buhk & Thielsch, 2015). So far, there have been studies on highly invasive hybrids (e.g. Gaskin & Schaal, 2002; Gaskin & Kazmer, 2009), however, the implications that future climate change will have on the frequency of hybridization events have hardly been evaluated so far.

Impact of Plant Invasions

Apart from processes leading to biological invasions understanding impacts of alien invaders at the species, community and ecosystem level has long been an important research focus. These impacts include, e.g. the decrease of local plant species abundance and diversity (Vilà *et al.*, 2006; Powell *et al.*, 2011; Vilà *et al.*, 2011), an altered N-cycle which affects plant performance and hence community structure (Liao *et al.*, 2008), impacts on human well-being (Essl *et al.*, 2015), or in general, any significant change in an ecological process or pattern (Pyšek *et al.*, 2012). These impacts differ in magnitude and direction among different ecosystems and also affect higher trophic levels (Vilà *et al.*, 2011). Different attempts in the past tried to classify mechanisms of impact, e.g. the International Union for Conservation of nature (IUCN) Global Invasive Species Database (GISD, <http://www.issg.org/database>) which distinguishes 13 different mechanisms along with their impact outcomes. To quantify and rank impacts on biodiversity and ecosystems in order to prioritise management responses Blackburn *et al.* (2014) developed a system based on the mechanisms of impact defined by the GISD. Due to its analogy to the accepted Red List approach for categorising extinction risk, their classification scheme could be readily integrated in policy (Blackburn *et al.*, 2014).

Invasion Hypotheses

The effectiveness of both, basic and applied research depends substantially on useful hypotheses that explain and predict biological invasions (Jeschke *et al.*, 2012). In fact, numerous such hypotheses have been formulated (Catford *et al.*, 2009). Recent evaluations by Jeschke *et al.* (2012) discovered substantial support (based on a global literature review on a subset of six hypotheses) for only a few of them: *Invasional meltdown hypothesis* – the presence of invasive species in an ecosystem facilitates further invasion by other alien species (e.g. Simberloff & Von Holle, 1999), *Novel weapon hypothesis* – advantage of invasive alien species due to a trait that is new in the resident community (e.g. Callaway & Ridenour, 2004), *Enemy release hypothesis* – invasion success due to the absence of natural enemies (e.g. Maron & Vilà, 2001; Liu & Stiling, 2006) and the *Tens rule* (only supported by studies dealing with plant species, however only partly) – where 10% of introduced species manage to establish self-containing populations in the wild and 10% of these naturalized species become invasive (Williamson & Brown, 1986; Jeschke & Strayer, 2005). Many other hypotheses have been proposed so far, however most of them are restricted to specific circumstances. Thus, studies testing the same hypothesis (or variations of these hypotheses) often reported contradictory results (Jeschke *et al.*, 2012; Moles *et al.*, 2012). To overcome those difficulties, different efforts have been made, formulating tools and frameworks for synthesizing invasion hypotheses (Catford *et al.*, 2009; Heger *et al.*, 2013). Heger and Jeschke (2014) classify hypotheses in a tree-like structure (broad hypotheses branching into more specific sub-hypotheses, also called hierarchy of hypotheses) and motivate scientists to validate and in case revise or replace not supported sub-hypotheses.

Ornamental Plant Species as Potential Future Invaders

The alien flora of a region consists of plant species that, both, have already become naturalized or invasive, or have been introduced to a region but have not escaped from cultivation yet. The latter group of species forms a massive pool of potential future invaders to the regional wild flora. “*In many regions, this pool is dominated by non-native plants used for public and domestic gardening (Hulme et al., 2008; Niinemets & Penuelas, 2008; Pergl et al., 2016). In Europe, for example, more than 16,000 species from more than 200 families are currently in cultivation for ornamental purposes, with many of them being alien to Europe (Cullen et al., 2011). Some of these non-native garden plant species have already become*

naturalized or invasive elsewhere in the world (van Kleunen et al., 2015) and can hence be considered particularly likely to do so in Europe too (Williamson, 1999)” (Klonner et al., 2017). With horticulture as the major pathway of introduction of alien garden plants (Hulme et al., 2008) and thus an important source of propagule pressure, future naturalization events might increase based on a continually increasing stock (Pergl et al., 2016) and a changing climate (see above). However, few studies have examined the effects that cultivation pattern and frequency may have on the spread of alien ornamentals, especially regarding a changing climate. Apart from a high propagule pressure through horticulture, ornamental plant species are given sufficient time to adapt to local conditions (Mack, 2000). Further they are offered abundant suitable habitats within urban areas, which can serve as starting points for successful naturalization of alien plant species (e.g. Chytrý et al., 2008a; Pyšek & Chytrý, 2014). However, little is known so far about which particular introduced ornamental species have the potential to naturalize or become invasive.

Along with the potential introduction of alien species through the pool of ornamental plant species other alien species, in particular aphids and scale insects (or in general pests, pathogens and parasites), can be carried along with their host plant (contamination; see also Hulme et al., 2008). Also, seeds, spores and eggs can be transported within soil and aggregates and thus provide a route for introduction of microorganism, animals and plants (Hulme et al., 2008).

Management of Invasive Alien Species

Management of invasive alien species is one of the key factors to mitigate negative impacts on native ecosystems, human well-being and economy (cf. Hulme, 2006). To be able to formulate management tools, information on the drivers of biological invasions (see above), their impacts on ecological patterns and processes (see above) as well as data on invasive alien species are needed. Concerns of society and science about alien species have led to improved documentation of their distributions, with inventories now being available for many regions (van Kleunen et al., 2015). Although inventories on certain taxa are still incomplete (invertebrates, microorganisms, less well-surveyed regions), vascular plants are well documented. Efforts to combine different inventories of alien species for large geographical regions and the most problematic invaders globally resulted in new databases, for example DAISIE (2017). Another global database recently established gives information on the

distribution of naturalized alien plant species, namely the Global Naturalised Alien Flora – GLONAF (van Kleunen *et al.*, 2015). Such databases are important for understanding global patterns of naturalization and can help making decisions for environmental managers.

The Convention on Biological Diversity (CBD, 2000) offers general strategic guidelines for the management of invasive alien species: prevention as the top priority response to invasive species (information, self-regulation and legislation, quarantine measures); followed by early detection (interception, monitoring and surveillance, removal), and management (eradication, containment, control) if prevention fails. While management efficiency decreases with time since introduction, costs are strongly increasing (Simberloff *et al.*, 2013). The development of management-, legislation instruments and codes of practice are crucial to prevent successful future invasions. This concerns not only a few but a lot of different sectors, responsible for the introduction of alien species (e.g. hunting federations, ornamental and pet trade, transport organizations, importers for commodities; Hulme *et al.*, 2008). A lot of such instruments have been developed so far managing introductions through trade in commodities (release, escape, contaminant – e.g. EU Birds and Habitat Directive, EU Zoo Animals Directive, Food and Agriculture Organization’s Code of conduct, EU regulation 1143/2014 on the management of invasive alien species). Still most of them are lacking effectiveness because they are not binding, not closely followed or not strictly executed (Hulme *et al.*, 2008).

Another important tool for the prevention of alien introductions are pathway risk assessments (see chapter ‘History’). Hulme (2009) propose using spatial data on climatic suitability, habitat and points of entry within demographic models to generate maps that highlight areas at risk of biological invasion. By including species dispersal parameters and measures of propagule pressure hotspots of invasion likelihood could be identified. Introduction or establishment of invasive species through horticulture will be challenging to manage based on the number of different actors (e.g. plant breeders, nurseries, wholesale suppliers, public retail outlets, public consumers) and their different motivations for market change and knowledge on invasive plant species (Humair *et al.*, 2014; Hulme *et al.*, 2017). Pre- and post-border risk assessments of invasive species, supported by industry codes of conduct and public education, all of which integrated along the horticulture industry supply-chain are effective policy instruments to cost-efficiently and environmental-desirably reduce plant invasions (Hulme *et al.*, 2017).

METHODS

Climatic Data

The Intergovernmental Panel on Climate Change (IPCC) is regularly publishing climatic data (i.e. temperature and precipitation data) within their assessment reports, the latest one being the Fifth Assessment Report (AR5; <https://www.ipcc.ch/report/ar5/>). The main objective of those assessment reports is to give information “*about the state of scientific, technical and socio-economic knowledge on climate change, its causes, potential impacts and response strategies*” (<https://www.ipcc.ch/activities/activities.shtml>). Within my dissertation I used three different IPCC5 scenarios, each based on a different Representative Concentration Pathway (RCP) that prescribes CO₂ concentration adding a distinct amount of radiative energy (W/m²) to the atmosphere by the year 2100: RCP2.6, RCP4.5 and RCP8.5. The scenarios are based on models that simulate climate, driven by global General Circulation Models and available at the Cordex portal (<http://www.euro-cordex.net>). To derive bioclimatic variables and a fine-gridded resolution I first had to process and downscale the spatial climatic data from Cordex (see ‘Appendix Manuscript 2’ in Dullinger *et al.*, 2017).

Modelling

To model the global realized climatic niche of species I used species distribution models (SDMs) which relate species occurrences with environmental data (Guisan & Zimmermann, 2000). Rising availability and access to vast sets of occurrence data has led to a widely use of SDMs in ecology (e.g. Thuiller *et al.*, 2005; Elith *et al.*, 2006; Broennimann & Guisan, 2008; Petitpierre *et al.*, 2012). Limitations to different kind of models (e.g. regression or machine learning techniques) led to the development of ensemble approaches, e.g. the BIOMOD2 platform (Thuiller *et al.*, 2009) in R (R-Core-Team, 2015) which I used within my studies. The package offers ten state-of-the-art modelling techniques to model a species’ relationship with its environment combined with the potential to make projections on e.g. future climate scenarios. By its nature BIOMOD can be used to only model binomial data, i.e. ‘presence/absence’ or ‘presence only’ (Thuiller *et al.*, 2009).

For one of my studies I used a coupled niche - demographic model (hybrid) to simulate demographic and dispersal processes of species (see ‘Methods’ in Appendix Manuscript 4). It

gives information on, both, occurrence and abundance of species by incorporating occurrence probabilities from SDMs along with dispersal- and demographic parameters (e.g. Dullinger *et al.*, 2012; Hülber *et al.*, 2016).

RESEARCH QUESTIONS

Within my first study I analysed plant species' trait profiles and their power for explaining invasiveness. Therefore I collected data on biological and distributional traits for 1402 species of the native, non-woody vascular plant flora of Austria. I then compared the subsets of species recorded and not recorded as invasive aliens anywhere in the world, respectively, first, with respect to the sampled traits using univariate and multiple regression models; and, second, with respect to their multidimensional trait diversity by calculating functional richness and dispersion metrics. The objective of this study was to expand the focus of 'invasive trait' – research by an evaluation about the variability or similarity of these two species groups in respect to their potential 'invasive traits'. Like in similar studies (Pyšek *et al.*, 1995; Lloret *et al.*, 2005; Küster *et al.*, 2008; Pyšek *et al.*, 2009; Pyšek *et al.*, 2014) I expected high predictive power at least of a subset of traits used. Moreover I hypothesized lower multidimensional trait variability among invasive than non-invasive species based on the idea that a distinct trait profile explains successful invasiveness.

In the following studies, I focused on ornamental plant species currently cultivated in Europe, as one of the most important source pools for potential future invaders. For the second study, species distribution models were used to assess changes in invasion risk from garden plants in Europe under a warming climate. We focused on a subset of European garden plants (Cullen *et al.*, 2011) that have already managed to become naturalized elsewhere in the world (Global Naturalized Alien Flora; <https://glonaf.org/>; e.g. van Kleunen *et al.*, 2015). Change in invasion risk was assessed by modelling current and future climatic niches of each species under three future climate change scenarios. We moreover integrated propagule pressure from gardening and urban landscaping into our calculations by combining areas with the highest number of naturalized species modelled (hotspots) with a weighted land-cover map accounting for the proportional area available for gardening (<http://www.eea.europa.eu/>).

With the third study I evaluated whether hybridization between garden plants and native (or other, already naturalized alien) species may increase under climate change. Therefore I used

species distribution maps of, both, potential invasive garden plants introduced in Europe (Dullinger *et al.*, 2017) and their native congeneric species to generate projections on their current and future potential climate niches. The set of potential invasive garden plants was restricted to congeners from genera with hybridization documented in the literature. I hypothesized that newly established garden plants have the potential to hybridize with their resident congeneric species (Ayres *et al.*, 2004; Bleeker *et al.*, 2007). I assessed the risk of hybridization by quantifying spatial overlap between the suitable areas of these two species groups under three different future climate change scenarios.

The fourth study used a large-scale simulation experiment to disentangle the relative effects of climate change and propagule pressure on possible future invasion success of garden plants in Europe until the end of the 21st century. The species were first and foremost potential invasive garden plants currently cultivated in Europe (see chapter 'Ornamental Plant Species as Potential Future Invaders' and Dullinger *et al.*, 2017) and second cultivated in experiments of collaborating research partners (e.g. Haeuser *et al.*, 2017). By coupling niche- and demographic models, escape and spatial spread of these species was simulated. Propagule pressure was regarded as 'human introduction effort' (Simberloff, 2009) and implemented by randomly selecting gardens across climatically suitable parts of Europe using six levels of cultivation frequency.

CONCLUSION

Analysing plant species' trait profiles and their power for explaining invasiveness by comparing subsets of species recorded and not recorded as invasive aliens anywhere in the world suggest moderate predictive ability of species traits (Klonner *et al.*, 2016). From the perspective of invasive species management and risk assessment, attempts to recognize invaders based on a specific trait profile could thus be associated with high uncertainties. Moreover, multidimensional variability in trait space is larger among invasive than among non-invasive species suggesting that invasion success is dependent on, both, abiotic and biotic context. As a corollary, defining 'critical' profiles might profit from regionally adapting these trait profiles such that they are complementary to resident native communities (i.e. species that are dissimilar to the native community in particular respects might be more successful). By having an advantage through a trait that is new to the native community, the results of this study support the '*Novel weapon hypothesis*' (see chapter 'Invasion Hypotheses' above).

Hotspots of naturalization risk defined by climatic suitability alone, or by a combination of climatic suitability and appropriate land cover, are in general projected to increase considerably in a warming Europe (Dullinger *et al.*, 2017). Even though the species distribution models show pronounced species-specific differences, a larger number of naturalized species could imply an increasing risk for potential invaders to be among them (Jeschke & Strayer, 2005). Overall the results of the second paper suggest that climate change may increase invasion risk from garden plants considerably and are in line with current expectations (Petitpierre *et al.*, 2016). They hence emphasize the need to raise public awareness of the potential problems associated with invasive garden plants among all actors in the horticulture supply chain, especially in the era of e-commerce (Humair *et al.*, 2015).

Species distribution models also suggest that future climate change does not seem to increase the mean geographic overlap of climatic ranges between potential invading ornamentals and their congeners in Europe (Klonner *et al.*, 2017). Hence the average risk that introduced garden plants will hybridize with their native congeneric species is unlikely to increase in the future. Nevertheless the species-specific results do include individual congener pairs showing strongly increasing overlap of suitable climatic ranges.

Niche-demographic models show that average areas simulated to be occupied by a subset of invading garden species at the end of the century are not larger but smaller than under a constant climate for two of three tested climate change scenarios (Klonner *et al. submitted manuscript*). This is likely due to a spatial displacement of suitable sites over the course of the century that drives transiently established populations to extinction and creates an invasion debt (Essl *et al.*, 2011) which may eventually be paid off when the climate should stabilize again. In addition, the simulations show that restrictions of use have a strong effect on the spread of non-native garden plants. Since this effect is non-linear, important benefits (i.e. effective reduction of naturalization events and spread of potentially invasive garden species) will only accrue if restrictions successfully reduce cultivation intensity to low levels. In order to ensure compliance legislation would need to integrate eligible policy instruments along the horticulture industry supply-chain (Hulme *et al.*, 2017).

As with every model, a suite of caveats apply to the results of, both, the species distribution models (see Dullinger *et al.*, 2017; Klonner *et al.*; 2017, Klonner *et al. submitted manuscript*) and the dispersal simulations (Klonner *et al. submitted manuscript*). First, I note that the

SDMs applied in these studies for estimating the species' realized climatic niche were fitted using 'Presence' data from an online database (Global Biodiversity Information Facility), which poses the risk of implicit biases that can affect geographic projections of the SDMs. Also, the realized niches of species often change in alien ranges (Early & Sax, 2014; Dellinger *et al.*, 2016), either due to differences in biotic environments or to rapid genetic adaptations (Prentis *et al.*, 2008). All of these factors may change modelling results importantly for individual species. However, by modelling the average effect of a set of species in each study inferences on the results should be robust. Second, for the study using niche-demographic models (Klonner *et al.* *submitted manuscript*) I note that I did not incorporate land use scenarios. Future climate and land use change will not be independent of each other, however, and strong warming may be associated with a spatial separation of land use (Spangenberg *et al.*, 2012) that facilitates invasion by non-native species in some parts, but reduces invasibility in other areas of Europe (Chytrý *et al.*, 2012). Apart from the possible effects of land use changes, caveats also relate to uncertainties in all the parameters fed into the niche-demographic model.

Overall the different modelling approaches suggest that climate change in general will likely facilitate future alien ornamental plant invasions in Europe based on the increase of their climatic suitable area. However, using niche-demographic models revealed that invasion success is unlikely to be fostered by different future climate change scenarios within the year 2090. Thus a successful invasion seems to not only depend on the area climatically suitable but on propagule pressure (i.e. human cultivation) foremost disregarding the climate change scenario. Even though horticulture is expected to facilitate plant invasions by filtering species based on traits that promote invasiveness (Drew *et al.*, 2010; Knapp *et al.*, 2012), their invasion success might also depend on a dissimilar trait set compared to native species (Knapp & Kühn, 2012; Klonner *et al.*, 2016), apart from other factors like decreased biotic resistance of resident communities (Eschtruth & Battles, 2009; Svenning & Sandel, 2013). In response to their potentially severe consequences, better management of biological invasions has become a priority of environmental policy (e.g. Council Regulation, 2014). The results here show that implementing applicable policy instruments (e.g. risk assessment protocols, import or sales bans, voluntary codes of conduct and consumer education; see Hulme *et al.*, 2017) along the horticultural supply chain to tackle invasions may in concert be effective despite imperfect compliance.

The studies conducted within this thesis are a contribution on the way towards developing models that can be used for risk assessment protocols identifying future alien plant invaders. and investigating mechanisms facilitating future biological invasions.

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RESEARCH ARTICLE

A Source Area Approach Demonstrates Moderate Predictive Ability but Pronounced Variability of Invasive Species Traits

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Abstract

The search for traits that make alien species invasive has mostly concentrated on comparing successful invaders and different comparison groups with respect to average trait values. By contrast, little attention has been paid to trait variability among invaders. Here, we combine an analysis of trait differences between invasive and non-invasive species with a comparison of multidimensional trait variability within these two species groups. We collected data on biological and distributional traits for 1402 species of the native, non-woody vascular plant flora of Austria. We then compared the subsets of species recorded and not recorded as invasive aliens anywhere in the world, respectively, first, with respect to the sampled traits using univariate and multiple regression models; and, second, with respect to their multidimensional trait diversity by calculating functional richness and dispersion metrics. Attributes related to competitiveness (strategy type, nitrogen indicator value), habitat use (agricultural and ruderal habitats, occurrence under the montane belt), and propagule pressure (frequency) were most closely associated with invasiveness. However, even the best multiple model, including interactions, only explained a moderate fraction of the differences in invasive success. In addition, multidimensional variability in trait space was even larger among invasive than among non-invasive species. This pronounced variability suggests that invasive success has a considerable idiosyncratic component and is probably highly context specific. We conclude that basing risk assessment protocols on species trait profiles will probably face hardly reducible uncertainties.

Introduction

The search for traits which define successful invaders is one of the fundamental issues in invasion biology [1–3]. Identifying such traits is not only of scientific interest but would also facilitate predictions about which species might cause ecological or socio-economic problems upon introduction and hence help to improve proactive management. Researchers have thus used various approaches to detect such ‘invasion traits’, or trait values, like the comparison of invasive and/or non-invasive alien species with native ones in the introduced ranges, or the

comparison of native species that have become invasive elsewhere in the world with those that have not (e.g. [3, 4, 5–9]). For plants, these efforts have shown that attributes like fast growth, maximum size and high dispersal abilities actually tend to be positively related to invasiveness (e.g. [7, 10]). However, the ability of these traits to discriminate invasive from non-invasive species was mostly moderate at best, and an undisputed set of attributes that clearly distinguish invaders has not yet emerged (e.g. [11]).

Inconsistencies among the traits under study, interactions among traits (e.g. [6]) as well as methodological differences (e.g. [10]) might have contributed to the mixed results achieved so far. In addition, the trait profile that makes a species invasive may not necessarily be the same in all contexts (e.g. [12]). As an example, recent studies comparing functional traits among invasive and co-occurring native species have demonstrated alien species are functionally distinct from native communities, i.e. they are functionally more distant from the native community than the native species are among themselves [13, 14, 15, 16]. These findings support the idea that successful invasion into native communities requires some ‘empty niches’ or parts of the trait space not already occupied by the resident species [16–18]. As native communities are diverse in their trait profiles, the attributes and attribute combinations that make species invasive may therefore be expected to vary too, either in different native communities within one region or in different regions that have different species pools.

Interestingly, however, studies searching for ‘invasion traits’ have so far concentrated on comparing mean trait values among successful invaders and different comparison groups but have rarely focused on trait variability within the group of invasive species. Individual studies hence show that the average invasive species differs, or does not differ, from the average species of some comparison group with respect to particular traits (e.g. [3, 10, 19]), but provide little information about the dispersion of trait values within the two groups. Knowledge of this variability is clearly of importance for our understanding of invasions as large variability likely implies complex and highly context-dependent causation while low variability indicates a predominant, generic impact of only few factors and processes on invasive success. Knowing this variability is also essential for managing and controlling plant invasions as risk assessment protocols are often based on traits [20] and the ability to recognize potential invaders from their trait profiles decreases with increasing variability of these traits. In other words, if trait variability among invaders is large, trait-based risk assessment protocols become less reliable because misclassifications of risk species will likely increase.

The objective of this study is to expand the focus of ‘invasive trait’-research by combining a screening for potential invasive traits or trait values with an evaluation of how uniform, or variable, the group of invasive species actually is with respect to these potential ‘invasion traits’ as compared to a non-invasive contrast group. Using the non-woody flora of Austria as a study system, we thereby apply a so-called source area approach [7, 17, 18] which focuses on a regional native species pool and compares species that have become invasive elsewhere with those that have not. As compared to the target area approach, i.e. the comparison of invaders and natives in the formers’ invaded ranges, this strategy reduces potential confounding effects of variable evolutionary predispositions [21–23] and differential reachability of invaded ranges for the members of a regional invader pool [7]. We expect that like in similar studies (e.g. [6, 7, 9, 24, 25]), our screening will demonstrate that invasive and non-invasive species differ with respect to at least a subset of traits. With respect to variability in these traits, by contrast, the lack of explicit studies constrains the formulation of clear hypotheses. However, the idea that potential invaders may be recognizable from their traits suggests that being a successful invader requires a certain, distinct trait profile, whereas no such constraints apply to the set of non-invasive species. Our working hypothesis is hence that trait variability is lower among invasive than among non-invasive species.

Methods

Data collection

Our analysis focused on the native, non-woody terrestrial vascular plant flora of Austria because knowledge about traits and habitat affiliation of this source area is consistent and relatively complete. The flora of Austria is rich in species as the country covers all the different biogeographical regions of Central Europe [26]. In addition, European regions are particularly suited for a source area approach as Europe has historically served as a main donor of invasive plants for the rest of the world [8, 27].

From the total set of vascular plants in Austria we restricted our analysis to the subset of terrestrial, non-woody spermatophytes, i.e. we excluded the life forms of phanerophytes, chamaephytes and hydrophytes as well as ferns, clubmosses and horsetails, because concentrating on particular taxonomically or ecologically defined plant groups or growth forms was suggested to be more promising when searching for invasion traits [3, 28]. Further, we removed all species not indigenous to Austria, both neophytes and archeophytes, to adhere to a source area approach in a strict sense. Moreover, we excluded 251 species from taxonomically insufficiently resolved, often apomictic genera like *Alchemilla* or *Hieracium* for which taxonomic treatment is likely to vary among different floras and invasive species lists. Finally, we could not consider species which were not represented in the trait databases used (see below). These successive reduction steps left us with a set of 1402 species.

For these 1402 species we searched for trait information in several different trait databases, namely the Ecological Flora Database [29], BiolFlor [30], LEDA [31], CLO-PLA [32], Fischer et al. [33] and Jäger et al. [34]. Based on considerations of both relevance (e.g. [6–9, 24, 25]) and data availability, we selected eight biological traits and differentiated the following trait states (1) Life form after Raunkiaer [30]: therophyte, geophyte, hemicryptophyte and hemiphaenaerophyte. (2) Life span as an indicator of generation time [30]: annual, biennial, perennial-pollakanthic, perennial-hapaxanthic. (3) Mating system as an indicator for uniparental reproduction [30]: allogamous, autogamous, mixed mating. (4) Pollen vector [30, 34]: wind, insects, self-pollination, cleistogamy. (5) Type of reproduction [30] according to [30]: by seed and vegetative, by seed, mostly by seed and rarely vegetative, mostly vegetative and rarely by seed, vegetative. (6) strategy type [30] according to [35]: competitors, competitors/ruderals, competitors/stress-tolerators, competitors/stress-tolerators/ruderals, ruderals, stress-tolerators, stress-tolerators/ruderals. (7) Ellenberg's N-indicator value as a proxy of the species' ability to exploit high nutrient values [30, 34]: 1 (nutrient poor) to 9 (nutrient rich). Ellenberg's N-indicator values represent an ordinal classification of plants according to the position of their realized [ecological niche](#) along a gradient of nutrient availability. Although they are based on expert judgement and refer to the species' niche optimum only, they have been proven reliable and useful ecological indicators in many studies (e.g. [36]). (8) maximum plant height, according to [33]: to homogenize data types, we discretized this only numeric trait into (ordered) factor levels (S1 Table). In addition, we collected data on traits related to abundance in the source area and to use as an ornamental species, both indicators of the likelihood of being transported elsewhere and hence of propagule pressure (e.g. [7]), and to habitat requirements: (9) frequency of species within Austria [33]: very rare, rare, dispersed, frequent, very frequent; (10) ornamental use in Austria [37]; (11) presence in agricultural/ruderal habitats [33]: yes/no; (12) presence around aquatic habitats [33]: yes/no; (13) presence below the montane belt as unintentional transportation indicator [33]: yes/no; (14) number of altitudinal belts as an indicator of climatic tolerance [33]. For statistical analysis these traits were classed into five groups: (A) life history (1–2), (B) reproduction (3–5), (C) competitiveness (6–8), (D) habitat use (11–14) and (E) propagule pressure (9–10). Table 1 gives an overview about which trait was assigned to

which trait group. We also recorded the family to which each species belongs to avoid possible evolutionary dependence among species in the statistical analyses [33]. As the response variable, i.e. as an indicator of which species of the Austrian flora are actually invasive somewhere else, we used the classification of a species as an environmental weed in the latest edition of Randall's Global Compendium of Weeds [38]. Weediness *sensu* this source implies a certain economic or environmental impact [39], i.e. a state of invasion that is going beyond mere establishment of a species as part of a regional flora, i.e. naturalization, and which is usually associated with considerable frequency/abundance. With reference to the state-classification of alien invasions by Richardson et al. [40], 'weediness' is hence close to the latest state named 'invasion' (in contrast to 'introduction' and 'naturalization'). In our analysis, we concentrate on this state of invasion because at the time of this study, Randall's Global Compendium of Weeds [36] was the only comprehensive global list of invasive species. Finally, all species not included in Randall [36] were classified as non-invasive.

Statistical analysis

As we did not have a phylogenetic tree of our species available, we used generalized linear mixed effects models (GLMMs) with a logistic link function and family as a grouping variable to explore which of the traits collected were significantly related to the probability of a species

Table 1. Traits and trait groups used to explain invasiveness.

| trait group | trait | effect | R ² | AIC |
|----------------------|--|------------------------|----------------|--------|
| intercept-only model | | - | 0 | 1426.3 |
| life history | | | 10.4 | 1369.7 |
| | life form | geophyte | 1.1 | 1425.8 |
| | life span | annual | 8.6 | 1383.5 |
| reproduction group | | | * | 1417.8 |
| | reproduction | seed veg | * | 1417.9 |
| | mating system | - | * | 1426.5 |
| | pollen vector | - | * | 1425.1 |
| competitiveness | | | 20.9 | 1298.9 |
| | strategy type | c in general, r, sr | 8.7 | 1375.0 |
| | N-value | with increasing rank | 14.2 | 1333.1 |
| | maximum plant height | with increasing height | 4.8 | 1398.4 |
| habitat use | | | 34.6 | 1199.8 |
| | occurrence in agricultural or ruderal habitats | yes | 13.0 | 1298.1 |
| | occurrence around aquatic habitats | yes | 1.9 | 1411.3 |
| | occurrence under the montane belt | yes | 29.2 | 1285.1 |
| | number of altitudinal belts | with increasing number | 2.7 | 1404.2 |
| propagule pressure | | | 12.7 | 1316.8 |
| | frequency | very frequent | 12.4 | 1327.3 |
| | ornamental use | yes | 2.7 | 1406.1 |

The table presents the traits tested, their combination to groups as well as the marginal R_e [42, 43] and Akaike information criterion (AIC) values of models using these traits and trait groups to explain invasiveness of Austrian non-woody vascular plants in other parts of the world. "Effect" indicates which trait levels promote invasiveness most strongly.

* R^2 cannot be calculated due to convergence problems.

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becoming invasive. To avoid overfitting the GLMMs by using 14 different traits with partly multiple factor levels, and all their possible interactions as predictors, we performed this search for 'invasion traits' in several steps. First, we fitted a univariate model for each of the 14 traits separately. Then, we combined the 14 traits into five groups ([Table 1](#)) and fitted multiple GLMMs for each particular group to assess the relative importance of the single traits within these groups when other traits of the same group are simultaneously accounted for (in these models we fitted a random intercept for plant family only, because model parameter search algorithms did not converge for more complex models). Finally, we summarized the variables in each group by the first axis of a multiple correspondence analysis (CA) and then searched for the most parsimonious model using all five CA-axes and all their possible two-way interactions as potential independent variables. We used CA because almost all of our variables contain trait information as categorical rather than numeric values. To make the fixed effect estimates of the axes from the five different CAs directly comparable we standardized all 1st-axis-values and tested them on collinearity before running the GLMMs. The most parsimonious model was then selected based on the Akaike information criterion (AIC). Since the AIC only provides an estimate of the relative fit of alternative models [41] we also calculated a marginal R^2 [42, 43] which describes the proportion of variance explained by the fixed factors of the GLMMs alone.

We used measures of functional diversity to assess the variability of invasive and non-invasive species in multidimensional trait space. The measurement of functional diversity of communities or species groups has recently undergone important progress and several distance-based functional diversity indices have been developed [44–46]. Among those, we selected two different ones which both measure the dispersion of species in trait space [46]: Functional Richness (FRic) is based on calculating convex hulls in multidimensional trait space, i.e. the minimum geometry that contains all species as an indicator of the volume of functional space occupied by a community [45]. FRic is hence a measure of multidimensional trait variability analogue to the range of values of a single variable. Functional Dispersion (FDis), by contrast, the mean distance of individual species to the centroid of all species in multidimensional trait space, is rather analogous to the variance of individual variables and hence less sensible to outliers than FRic [46]. Here, we calculated both indices by means of the FD-package in R [46]. The FD-package computes diversity indices from semi-quantitative and qualitative variables by replacing them by the axes of a principle coordinate analysis (PCoA) of a Gower dissimilarity matrix. Negative eigenvalues of PCoA-axes were handled by adding the minimum constant to the distances that makes all eigenvalues positive [47].

To assess if multidimensional trait variability among invasive alien species is smaller, equal to, or even higher than among non-invasive species we compared the FRic and FDis of the subset of species classified as invasive against those of a re-sample (without replacement) of an equal number of species from the total pool of all 1402 species. We repeated re-sampling 1000 times and evaluated if the empirical values of the invasive group are within the 0.95 confidence interval of the 1000 values of these resamples. We conducted this comparison with FRic and FDis calculated for the overall set of traits as well as for the subsets of traits found to distinguish invasive and non-invasive species in the previous analyses [48].

All statistical analyses were conducted in R 3.1.1 [49] using contributed packages lme4 1.1–6 [50], ade4 1.6–2 [51], piecewiseSEM 1.0.0 [52] and FD 1.0–11 [53].

Results

From the set of 1402 species 305 were classified as invasive elsewhere whereas 1097 were not considered to be invasive aliens anywhere in the world ([S3 Table](#)).

Which traits are most related to invasiveness?

The traits analysed showed different effects on the species' probability of becoming invasive outside their native range (Table 1 and S1 Fig). Traits indicating the use of particular habitats were most useful for discriminating invasives, in particular "occurrence below the montane belt" and "occurrence in agricultural or ruderal habitats". Among the other variables tested preference of sites with high nitrogen availability, frequency of the species within Austria, and a competitive or ruderal strategy correlated with invasiveness most clearly. However, none of the single trait models was convincingly distinguishing native species, which are either invasive or non-invasive elsewhere. Summarizing traits into groups and calculating multiple models demonstrated that the trait group 'reproduction' explained invasiveness worst (highest AIC value), while the trait groups 'habitat use' and 'competitiveness' had highest predictive abilities (Table 1).

Among models which used CA axes (corresponding to the different trait groups) the best one not including interactions explained 40.35% of the variance (see AIC-values in S2 Table). All groups except 'reproduction' were included in this model with 'habitat use' having by far the strongest and 'life history' the weakest effects. Including interactions further improved models slightly. The best model had a marginal $R^2 = 45.93\%$, but reduced the AIC by only $\Delta AIC = 4.2$ as compared to the best model without interactions; it also included only one of all possible interactions, namely the one among competitiveness and habitat use (Table 2 and S2 Table).

Variation in trait space

According to the previous analyses the traits 'life form', 'mating system' and 'pollen vector' did not improve a random effects model (= an intercept-only model) by a $\Delta AIC > 2$ and were hence considered not useful to distinguish invasive and non-invasive species [48]. The trait 'ornamental use' was not included in this analysis since we did not expect any relationship with functional diversity. When measured by FRic the variation in multidimensional trait space among the species classified as invasive was in general even larger than the variation of a random sample of native plants from the Austrian source pool (Fig 1). This result held independent of whether all traits or only those useful to distinguish invasives were used for calculating FRic (Fig 1A). Using FDis, which is less sensitive to outliers, yields similar results: the functional dispersion was larger for invasive species when calculated either on the whole set of traits or on the set of 'invasive traits' (Fig 1B).

Discussion

In summary, our results demonstrate that, as expected, most of the analysed traits are actually related to their probability of becoming invasive outside their native range to a certain extent.

Table 2. Best GLMM to explain invasiveness of Austrian non-woody vascular plants in other parts of the world.

| | estimate | std. error | z-value | | p-value |
|-----------------------------|----------|------------|---------|------------------------|---------|
| AIC = 1128.8; $R^2 = 45.92$ | | | | | |
| life history | 0.28 | 0.08 | 3.63 | 2.85×10^{-4} | *** |
| competitiveness | 0.46 | 0.09 | -4.93 | 8.21×10^{-7} | *** |
| habitat use | 1.31 | 0.13 | -9.88 | 2.00×10^{-16} | *** |
| propagule pressure | 0.51 | 0.08 | 6.49 | 8.52×10^{-11} | *** |
| competitiveness:habitat use | 0.24 | 0.10 | -2.51 | 1.21×10^{-2} | * |

Traits represent first axes of correspondence analyses of the respective trait groups (Table 1) which were standardized before running the GLMMs. Best models were selected based on the Akaike information criterion (AIC) from all possible candidate models (S2 Table). The model's corresponding marginal R^2 value [42, 43] and Akaike Information Criterion are also shown. *, *** give information on the p-values significance.

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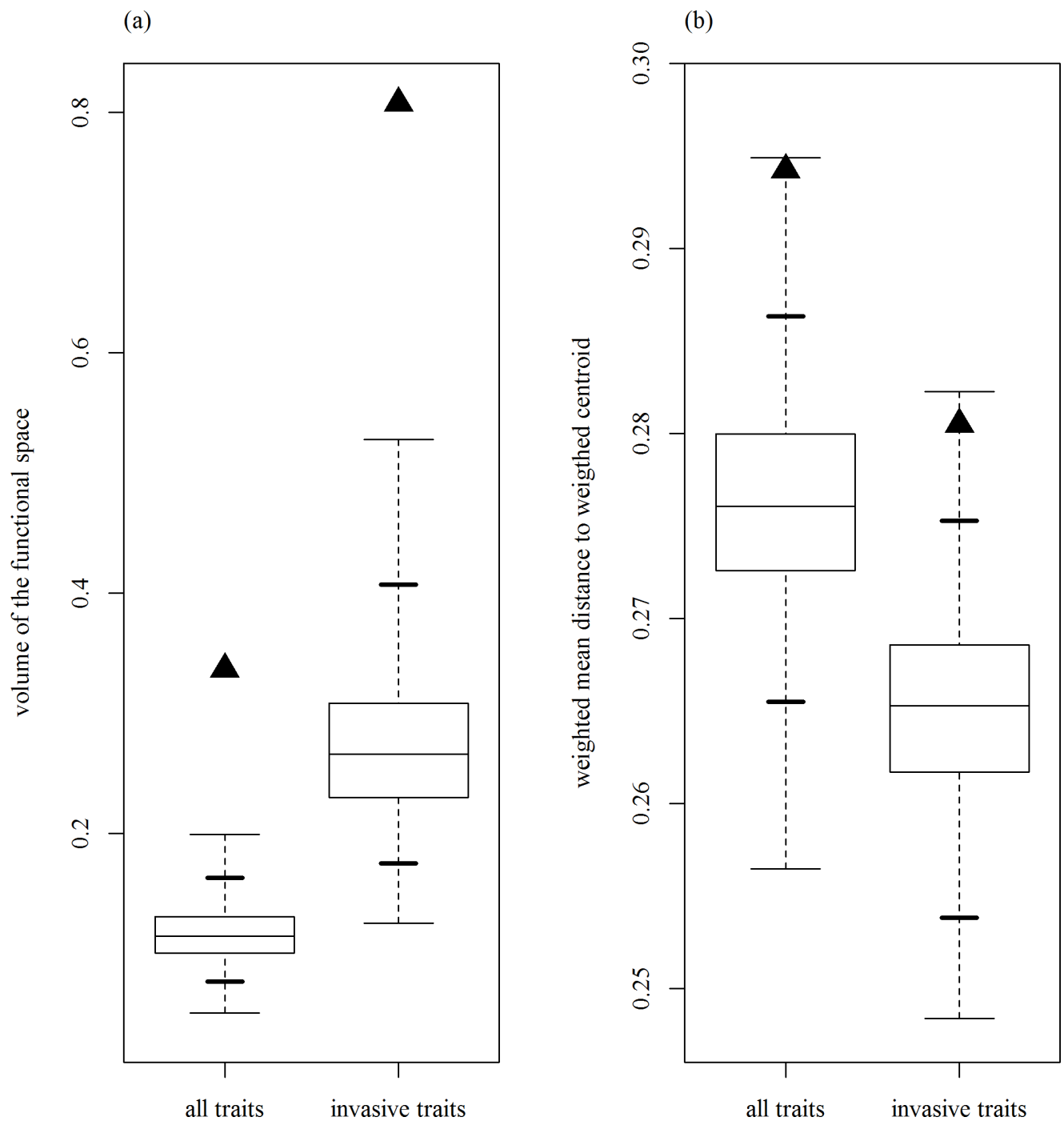


Fig 1. Comparison of functional diversity indices among Austrian plants that are either invasive or not invasive elsewhere in the world. Black triangles symbolize the diversity index values calculated for the group of the 305 invasive plants. The boxplots represent the range of index values calculated for 1000 equally large re-samples from the whole pool of 1402 native species with the bold lines indicating the 0.95 confidence interval of these re-sample based values. Panel (a) represents results for Functional Richness (defined by the volume of the functional space) and panel (b) results for Functional Dispersion (defined by the mean distance in multidimensional trait space of individual species to the centroid of all species), respectively. Label 'all traits' give results calculated with the total set of collected traits, label 'invasive traits' calculations based on traits that proved useful to distinguish invasive and non-invasive species in the preceding analyses.

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Nevertheless, neither a single trait group nor their combination to one best model allowed for distinguishing invasive aliens with high reliability. In addition, and against our ‘working hypothesis’, we found that multidimensional variability in trait space is even more pronounced among invasive as among non-invasive plants.

Single trait effects

Our single trait analyses suggest that attributes of the species related to the use of specific habitats are among those most closely correlated to invasion success. In particular, plants which occur in agricultural and ruderal habitats are more likely to become invasive outside their native ranges. This correlation is probably driven by a combination of different factors with both pre-adaptation to sites particularly sensitive to invasions and propagule pressure playing key roles. On the one hand, agricultural and ruderal habitats are not only donor but also important recipient habitats of invaders (e.g. [8, 54, 55]) because, first, they are characterized by a high frequency of human disturbance events which keep competitive resistance against invasion low; and, second, because the surroundings of sites of introduction, and hence of high propagule pressure, are usually characterized by high human population density and intense land use, i.e. by a high incidence of ruderal and agricultural habitats (e.g. [56]). As a corollary, biological traits that are selected by the conditions prevailing in these habitats, such as short life span, efficient reproduction by seeds and, in general, a ruderal life history strategy [35] are also among those positively associated with invasion success in our as well as in other studies (e.g. [7, 17, 57]). On the other hand, the preference of agricultural and ruderal habitats in the native range might also be related to invasions because of a sampling effect—the higher human population and the more intense usage also increase the likelihood of exportation of propagules from these areas. This sampling effect, and hence propagule pressure, is clearly also driving the influence of species frequency on invasiveness (cf. [7]). By contrast, the fact that invasive species mainly grow in lowlands of their Austrian range might again have to do with both matching between native and invaded habitats and the concentration of introduction foci in lowland areas. Alexander et al. [56] have, for example, suggested that the bias of introduction events towards lowland areas might prevent first establishment of mountain plants in the introduced range and hence also their expansion into more suitable sites at higher elevations—and thus explain why mountains are less affected by invasions. Moreover, propagules of mountain plants probably have a lower uptake probability, at least in temperate regions [8, 27], i.e. they are more rarely transported by humans over long distances because mountains are more sparsely populated and less intensively used than lowland areas. Thus, mountain species are probably not only less likely to establish at (typical) introduction sites, they are also less likely to reach them.

Regarding biological traits we found those associated with high competitive ability, like maximum height and ability to exploit high nutrient supply levels to show strongest correlation with invasiveness. These results are in line with findings of experimental studies which showed invasive species to usually have high growth rates, tall size as well as high leaf and shoot allocation [3, 58–60]. Similar to Pyšek et al. [7] our data suggest that in this context invasiveness is less conferred by a particular trait like the often studied plant height (e.g. [19, 61, 62])—which has a significant, but relatively low effect on Austrian species becoming invasive elsewhere—but rather by a life history syndrome composed of a whole set of traits [6]. The ability to translate high nutrient input into fast growth is an important component of this trait combination which distinguishes C from S-strategists in particular [35]. The clear effect of N-indicator values on invasiveness hence fits well with the one of the competitive strategy type and is in line with other studies demonstrating that invasive species usually prefer sites with high nitrogen

supply rates [24, 59, 63]. In addition, plants introduced into new regions often shift towards faster growth strategies because of a reduction in the top-down constraint imposed by herbivores [64], and thus may effectively use high nitrogen supplies. However, high nutrient availability is also characteristic for ruderal and agricultural habitats and the relative roles of preadaptation, or habitat matching, and propagule pressure in driving the N-effect on invasiveness are hence again hard to disentangle.

For some of the traits tested expected relationships with invasiveness were actually not detected at all. For example, autogamous species should have an advantage during invasions because they do not depend on mating partners for establishment and spread [65, 66], but we did not find any significant effect of mating systems on the invasiveness of Austrian plants. We hypothesize that the lack of such an effect is partly due to our focus on invasiveness, i.e. on the latest stage of the invasion process, while the advantage of self-compatibility may particularly be relevant during early stages when populations of the invading species are still small and Allee-effects might play a prominent role [67]. Additionally, species may also change traits from native to invasive ranges. In some species, invasions have for example been associated with shifting investment from sexual to vegetative reproduction, or vice versa (e.g. [68]). Such shifts potentially mask the relationship between traits and invasive success and must go undetected with a source area approach. This possible shortcoming underlines that reaching unambiguous conclusions in invasion biology will often require the combination of different approaches [3].

Multiple trait models

Combining traits into groups and building a multiple model that integrated all these groups improved the distinction between invasive and non-invasive plants considerably. The best model, including interactions, explained ~ 46% of the variance. This value puts our model closer to the one reported by Pyšek et al. ([7]: 43% of variance explained by their best model) than to those reported by Küster et al. ([6]: c. 25% of variance explained by their best model). Both studies analyzed the relative importance of different trait sets for species' invasion success but either used a source area [6] or a target area approach [7]. Further Pyšek et al. [7] address the stage-structure of the invasion process while Küster et al. [6] highlight the importance of incorporating trait interactions when testing for traits that promote invasion. The lower success in the latter study might result from the fact that Küster et al. [6] used a target area instead of a source area approach and that they focused on all naturalized, and not only on invasive species. Actually, the effect of biological traits on naturalization rather than invasiveness has been found to be considerably lower by Pyšek et al. [7], too. Moreover, in agreement to Küster et al. [6, 69], interactions among traits had an effect on invasiveness in our models, although this effect was rather weak. We speculate that these interactions might be even more relevant when focusing on a specific region of introduction, like the target area approach does (e.g. [6]), than when pooling invasiveness across all the different adventive ranges of a large set of species like in the source area approach. We assume this to be the case because, with the latter focus, interactions of traits with highly variable abiotic and biotic conditions in the introduced ranges may become much more important than interactions among the traits themselves (cf. [24, 25] and discussion below).

Multidimensional trait variability

Following the idea that invasive species may be identifiable from distinct trait profiles, we had expected that trait variability among invasive species is smaller than among non-invasives. Our results did not corroborate this expectation. By contrast, invasive species appeared even more variable in their trait profiles than the non-invasive contrast group. One explanation of this result may be that important determinants of invasive success, which interact with the

evaluated traits, are missing from the model. Among these determinants, quantitative data on introduction efforts or propagule pressure and hence establishment opportunities, might be particularly important [70]. In addition, high variability in 'invasive traits' is also consistent with the idea that traits conferring invasive success depend on specific ecological settings in the recipient area. With respect to different habitat types such specificity of 'invasion traits' has already been discussed (e.g. [3, 24]). As an example, successful invaders might rather be ruderal strategists sensu Grime [35] when spreading into highly disturbed agricultural or urban habitats while a competitive strategy might be more promising when disturbance frequency is lower, like in many semi-natural or natural habitats. Even an S-strategy might be helpful in particular cases, e.g. when species are invading regions characterized by cold temperatures or low water availability. Recent studies have moreover shown that there is a strong negative correlation among the trait profiles of invaded communities and the attributes necessary to invade these communities at local scales [13, 14], while at larger spatial scales this correlation may switch into a positive one [16]. These results strongly suggest that, for becoming a successful invader, differences in trait profiles from those prevailing in the native communities might be more important than specific trait values *per se*. In a source area approach, species invasive all over the world are simultaneously considered. The variation among recipient areas and invaded communities is hence large. In light of these recent studies which emphasize the strong context dependence of invasive success the pronounced trait variability that we found among invaders in our source areas approach appears hence less surprising.

From the perspective of invasive species management and risk assessment, our results imply that attempts to recognize potential invaders based on traits will remain challenging because high variability is likely associated with considerable 'error rates'. If this variability is, as hypothesized, at least partly due to (dis)similarities of successful invaders with native communities an appropriately differentiated approach might actually be promising. In other words, our results question the existence of a distinct trait profile that makes species invasive independent of the abiotic and biotic context. Further they suggest adapting critical trait profiles in risk assessment protocols to the particular environmental conditions and trait profiles of resident native communities as far as possible.

Supporting Information

S1 Fig. Proportions of plants that are invasive/ not invasive in other parts of the world.
(PDF)

S1 Table. Categorization of 'maximum plant height'.
(PDF)

S2 Table. Evaluation of candidate generalized linear mixed effect models based on the Akaike information criterion.
(PDF)

S3 Table. List of study species analysed and their status of being invasive somewhere outside their native distribution.
(PDF)

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Author Contributions

Conceived and designed the experiments: SD FE. Analyzed the data: GK SF. Wrote the paper: GK SD. Compiled the data: SF.

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RESEARCH
PAPER

Climate change will increase the naturalization risk from garden plants in Europe

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ABSTRACT

Aim Plant invasions often follow initial introduction with a considerable delay. The current non-native flora of a region may hence contain species that are not yet naturalized but may become so in the future, especially if climate change lifts limitations on species spread. In Europe, non-native garden plants represent a huge pool of potential future invaders. Here, we evaluate the naturalization risk from this species pool and how it may change under a warmer climate.

Location Europe.

Methods We selected all species naturalized anywhere in the world but not yet in Europe from the set of non-native European garden plants. For this subset of 783 species, we used species distribution models to assess their potential European ranges under different scenarios of climate change. Moreover, we defined geographical hotspots of naturalization risk from those species by combining projections of climatic suitability with maps of the area available for ornamental plant cultivation.

Results Under current climate, 165 species would already find suitable conditions in > 5% of Europe. Although climate change substantially increases the potential range of many species, there are also some that are predicted to lose climatically suitable area under a changing climate, particularly species native to boreal and Mediterranean biomes. Overall, hotspots of naturalization risk defined by climatic suitability alone, or by a combination of climatic suitability and appropriate land cover, are projected to increase by up to 102% or 64%, respectively.

Main conclusions Our results suggest that the risk of naturalization of European garden plants will increase with warming climate, and thus it is very likely that the risk of negative impacts from invasion by these plants will also grow. It is therefore crucial to increase awareness of the possibility of biological invasions among horticulturalists, particularly in the face of a warming climate.

Keywords

Alien species, horticulture, hotspot analysis, invasion debt, ornamental plants, species distribution model.

INTRODUCTION

Biological invasions can be conceptualized as a series of consecutive stages – from transport out of the native range to introduction into a new territory, naturalization or establishment of self-sustaining populations, and spread across the introduced range (e.g. Blackburn *et al.*, 2011). The term ‘invasive’ or ‘invader’ is thereby commonly reserved for species that have rapidly spread into multiple sites across a large area. To pass on to the next stage a species has to overcome specific barriers to its survival, establishment and spread. Whether and how fast a species manages to pass these barriers depends on a number of interacting factors that can be grouped into those relating to anthropogenic propagule pressure, physical conditions of the recipient area and biotic traits of the invader itself as well as of the invaded communities (Catford *et al.*, 2009). As a result of these consecutive filters, the number of species at each stage diminishes (Williamson & Fitter, 1996), and, even for eventually successful invaders, extensive time lags may separate first introduction, naturalization and subsequent spread (Essl *et al.*, 2011).

As climatic suitability of the new territory is particularly crucial for naturalization and spread (Catford *et al.*, 2009), expected climate change may importantly modify the number and identity of already introduced species able to pass to these subsequent invasion stages. Indeed, many examples have already been documented of alien species that have naturalized and/or started to spread in a region because recent warming trends have lifted former climatic limitations (Walther *et al.*, 2009). Predicting which species from a given pool of non-natives might actually benefit from upcoming climate warming, and where these species might become naturalized or invasive in the future, would provide a valuable basis for proactive management (Bradley *et al.*, 2012). So far, however, research efforts have concentrated on potential range expansions of species that have already become harmful (e.g. O’Donnell *et al.*, 2012; Bellard *et al.*, 2013) or at least naturalized (Duursma *et al.*, 2013) in the recipient area. These pre-selections exclude potentially large numbers of species introduced but not yet naturalized or invasive, which make up the pending invasion debt of a region (Essl *et al.*, 2011).

Alien species are introduced to recipient areas via different pathways (Hulme *et al.*, 2008). For vascular plants, intentional introduction for ornamental use has been identified as the major pathway world-wide (Hulme *et al.*, 2008). In Europe, for example, more than 16,000 species from more than 200 families are currently in cultivation for ornamental purposes (Cullen *et al.*, 2011). Public and domestic gardens thus contain the greatest pool of non-native plants on the continent (Niinemets & Peñuelas, 2008). The chance that in a warming Europe future invaders will primarily emerge from this pool is further increased by the fact that garden plants are often cultivated beyond the climatic limits of their natural populations and hence may get ‘a head start on climate change’ (Van der Veken *et al.*, 2008). In addition,

horticulture often selects for traits that also promote naturalization and spread, such as rapid growth, early and prolific reproduction and disease resistance (Mack, 2000; Pemberton & Liu, 2009; Chrobok *et al.*, 2011).

It remains hard to predict which particular species from the pool of introduced garden plants will actually manage to naturalize or even become invasive. What we do know, however, is: (1) species that have already managed to become naturalized somewhere in the world are more likely to escape from cultivation in other regions too (Williamson, 1999); and (2) that climate matching between native and introduced range is one of the few factors that consistently predicts invasion success across taxonomic groups and regions (Thuiller *et al.*, 2005; Hayes & Barry, 2008). Using these two ‘filters’ should hence help to at least select a subset of species with a higher risk of future naturalization and spread.

Here, we follow this rationale and explore whether the naturalization risk from currently cultivated garden plants will increase under a warmer climate in Europe. In essence, we first define the pool of non-native garden plants that have already naturalized as aliens somewhere outside of the continent, but not in Europe itself. Second, we parameterize species distribution models and use them to assess to what extent these species would already find suitable conditions for naturalization under the current climate and whether potential alien ranges would increase, on average, under three scenarios of climate warming. Third, we combine predictions for individual species into a ‘hotspot analysis’ (O’Donnell *et al.*, 2012; Bellard *et al.*, 2013) to identify areas with the highest numbers of potential future invaders under both current and future climatic conditions. Finally, we overlay these climatic hotspot maps with a weighted land-cover map accounting for the amount of potential ornamental planting area of each land-cover class (EEA, 2000) as an indicator of generic propagule pressure from gardening and urban landscaping.

METHODS

Data

Species selection and data

We selected from the European Garden Flora (EGF; Cullen *et al.*, 2011) all vascular plant species not native to Europe. (The EGF is the most comprehensive encyclopaedia of ornamental plants in Europe.) From this pool of species, we selected those which have successfully naturalized somewhere outside Europe but not yet anywhere in Europe, based on the Global Naturalized Alien Flora (GloNAF; van Kleunen *et al.*, 2015), a newly established global alien plant species distribution database which contains lists of naturalized alien plants in more than 850 regions covering 83% of the world’s terrestrial area. Cultivated taxa flagged as varieties or subspecies in the EGF were excluded to avoid overestimation when modelling the niches of the respective species. Moreover, we did not consider any taxa marked in the EGF as hybrids.

For this species subset, we then collated distribution data from the Global Biodiversity Information Facility (GBIF, <http://www.gbif.org>) using the *rgbif* library in R (Chamberlain *et al.*, 2015). All species were cross-checked for synonyms using The Plant List (<http://www.theplantlist.org>). Duplicates (i.e. multiple occurrences within $10' \times 10'$ grid cells) and obviously erroneous records, i.e. those on an ocean surface, were removed. After these cleaning steps, we retained 783 species with more than 50 occurrences irrespective of whether these stem from the species' native or non-native ranges (Gallien *et al.*, 2010; see Appendix S1 in Supporting Information).

Climate data

To characterize present-day climate, we used climatic data (averaged for the baseline period 1950–2000) from the WorldClim database (Hijmans *et al.*, 2005, www.worldclim.org) at a $10'$ resolution. From the 19 bioclimatic variables provided by WorldClim, we selected six which, in combination, represent a range of regional temperature and precipitation conditions together with an estimate of seasonal variability, and which are known to influence species distributions (Root *et al.*, 2003): (1) temperature seasonality, (2) maximum temperature of the warmest month, (3) minimum temperature of the coldest month, (4) precipitation seasonality, (5) precipitation of the wettest quarter and (6) precipitation of the driest quarter. Correlations (Pearson's *r*) among these variables were < 0.75 throughout and the impact of multicollinearity on model projections should hence be negligible (Dormann *et al.*, 2013).

Future climate was characterized by three different IPCC5 scenarios from the new Representative Concentration Pathways family: RCP2.6 ('mild' scenario), RCP4.5 ('intermediate' scenario) and RCP8.5 ('severe' scenario). Based on climatic models available at the Cordex portal (<http://www.euro-cordex.net>), we calculated mean predicted values of the six selected bioclimatic variables for the years 2050–2100 under these three scenarios (for detailed model selection and down-scaling procedure see Appendix S2).

Land-cover data

For the calculation of land-cover weighted risk maps, we used CORINE land-cover (CLC) data at a resolution of 100 m (EEA, 2000). The CLC land-cover classes were weighted by the estimated proportional area available for ornamental plant cultivation according to the descriptions in EEA (2000; cf. Chytrý *et al.*, 2009, for a similar approach). To safeguard against rating errors, we used three different weighting schemes, i.e. three different estimates of this proportional area per land-cover class (see Appendix S3 for details). In all three schemes, the highest weights were given to classes including private and public garden spaces (e.g. green urban areas). Within each scheme, we subsequently calculated the area-weighted means of these proportions for each $10' \times 10'$ raster cell.

Species distribution models

Model parameterization and evaluation

We modelled the global realized climatic niche of each species by combining available occurrence data with current climatic data within the *biomod2* platform (Thuiller *et al.*, 2009) in R (R Development Core Team, 2014). The four modelling algorithms used were: generalized linear model (GLM), general additive model (GAM), boosted regression tree (BRT) and random forest (RF). Since those algorithms require presence and absence data, but GBIF provides just 'presence-only' information, we generated 'pseudo-absences' following the recommendations of Barbet-Massin *et al.* (2012): for the regression technique models (GLM and GAM), we used 10,000 randomly distributed absences, and for machine-learning technique models (BRT and RF), we used a number of pseudo-absences equal to the number of occurrences found in GBIF and selected outside a radius of 200 km around these occurrences. For the latter approach, pseudo-absence generation, and hence model calibration, was repeated ten times per species to ensure that selected pseudo-absences did not bias the final predictions. For all models, the weighted sum of presences equalled the weighted sum of pseudo-absences. The predictive performance of the models was evaluated by means of the true skill statistic (TSS; Allouche *et al.*, 2006) based on a repeated (three times) split-sampling approach in which models were calibrated with 80% of the data and evaluated over the remaining 20%.

Model projections

Calibrated models were used to project the climatically suitable area for each species in Europe under current and possible future climatic conditions by means of an ensemble forecast approach (Araújo & New, 2007). As pseudo-absence generation differed between the two groups of models, we generated two separate ensemble predictions for each species, one from a combination of GLM and GAM, and one from a combination of BRT and RF models. In other words, the model projections from the repeated split-sampling approach (and from the repeated pseudo-absence selection in the case of BRT and RF) were aggregated to a weighted mean of projections. The contribution of each model to the ensemble forecast was weighted according to its TSS score. Models with a TSS score < 0.5 were excluded from building projections (see Appendix S4 for full information on model performance). The two probabilistic ensemble forecasts were translated into two binary maps using the value that maximizes the TSS score as the threshold for distinguishing presence and absence predictions. The two binary maps were then combined to a final consensus map where a $10'$ cell was defined to be suitable for a species (under a particular climate scenario) only if both binary ensemble layers predicted its presence. The latter decision rule makes the projections conservative, i.e. the extent of climatically suitable habitat is likely to be under- rather than overestimated.

To assess whether potential alien ranges of the 783 species will, on average, increase, decrease or remain constant in Europe under future climates, we compared SDM projections under current and future climates in terms of the number of cells predicted to be suitable for these species. As the distribution of these numbers was highly skewed, with an excess of zeros, we used a permutation test to evaluate the significance of differences: for each species, we randomly reshuffled the number of cells predicted to occur under current conditions and the future scenario, respectively, and calculated the difference (cells in the future scenario minus cells under current conditions). This calculation was done 1000 times, resulting in a vector of 1000 mean differences among the 783 species, which is normally distributed and centred around zero. Finally, we assessed if the actually observed difference was within or outside the central 95 or 99.9% of the simulated differences.

To analyse whether possible increases or decreases of alien ranges under climate change might depend on a species' biogeographical origin, we assigned the native regions of our study species to the nine climatically defined zonobiomes distinguished by Walter & Breckle (1991). Native regions were available for 704 of the 783 species in the GRIN database (<http://www.ars-grin.gov/>). Where native regions were assigned to more than one zonobiome, species were assigned to all of these zonobiomes. Finally, we re-did the same permutation tests as described above for the subset of species of each zonobiome separately.

Hotspot analysis and risk maps

For each climatic scenario, final binary consensus maps of all 783 species were stacked. From this overlay, we calculated for each 10' grid cell (*c.* 220 km² at latitude 50° N) the number of species that would find suitable climatic conditions there. We defined potential naturalization hotspots as the 10% of cells that provide a suitable climate to the highest numbers of species. To depict potential contraction or expansion of hotspots, we mapped the relative change in the areal extent of hotspots in comparison with the current climatic situation by applying the top 10% cut-off value (*i.e.* the number of species that separates the top 10% of the grid cells from the rest) determined under current conditions to the future climatic scenarios, too.

The hotspot maps represent the number of species that are predicted to be able to naturalize in particular regions (10' grid cells) based on their climatic requirements alone. Actual naturalization risk, however, also depends on the spatially variable amount of potential ornamental planting area. To create risk maps, we hence combined the stacked binary projections of the 783 species with each of the three weighted CORINE land-cover maps by multiplying the number of potential invaders by the area available for ornamental plant cultivation. We again defined hotspots of naturalization risk as the 10% of cells with the highest such multiplied values. The three resulting risk maps, one per weighting scheme of land-cover classes, were similar, but differed in some details (*cf.* Appendix S5).

We hence created a final consensus map where hotspots of naturalization risk were defined as those cells flagged as such by at least two of the three alternative risk maps.

RESULTS

Model projections and hotspot analysis

For 455 (*c.* 58%) of the 783 species included in our analysis, there is already a certain amount of suitable habitat (> 100 cells) in Europe under current climatic conditions. The number of suitable grid cells varies considerably among species (minimum 0, maximum 18,059, *i.e.* *c.* 58% of Europe), but is already > 1600 cells (*c.* 5% of Europe) for 21% of the species (165 species). Per raster cell, the number of species predicted to encounter suitable climatic conditions ranges between 0 and 305 (Fig. 1a). Northern and eastern Europe currently appear least suitable and western and southern Europe most suitable for our study species.

Under a warmer climate, both the mean potential range size per study species (Fig. 2a–c) and the number of species finding particularly large climatically suitable ranges in Europe (Fig. 2d) increase. Enlargement of mean potential range sizes is greater the more pronounced the climate-change scenario (Fig. 3). However, not all the analysed species are predicted to profit from warmer climates. The modelled species pool is separated into those likely to gain and those which will lose climatically suitable area in a warmer Europe. The gap between these two groups becomes, again, the more pronounced the more severe the climatic scenario (Fig. 2a–c).

Separating species according to their biogeographical origin demonstrates that those native to nemoral and laurophyllous zonobiomes profit most, especially under the most severe scenario, while those native to boreal and Mediterranean zonobiomes benefit least or even decrease in mean range size under the most severe climate scenario (Fig. 3). However, at least some species from any zonobiome show particularly strong reduction or enlargement of potential range size under each climate scenario, with pronounced losers being particularly frequent among boreal, nemoral and Mediterranean species (Fig. 2, Appendix S6).

Similar to species, geographical regions are also separated into those gaining and losing potential invaders with a warming climate (Fig. 1b–d). Gains are particularly pronounced in the north-western and eastern parts of Europe while the southern Atlantic and most of the Mediterranean coast are predicted to be suitable for a lower number of ornamentals under future climates.

Under current climatic conditions 10% of Europe is climatically suitable for at least 70 from our pool of 783 species. These climatic hotspots are clustered along the Atlantic coast of Portugal, Spain, France and the southern British Isles as well as along the Mediterranean coast of the Balkan Peninsula and in southern central Europe (Fig. 4a). Under future climates, the hotspot area is predicted to grow, *i.e.* the area that provides climatically suitable habitat to ≥ 70 species will become larger by 62% under RCP2.6, by 75% under RCP4.5

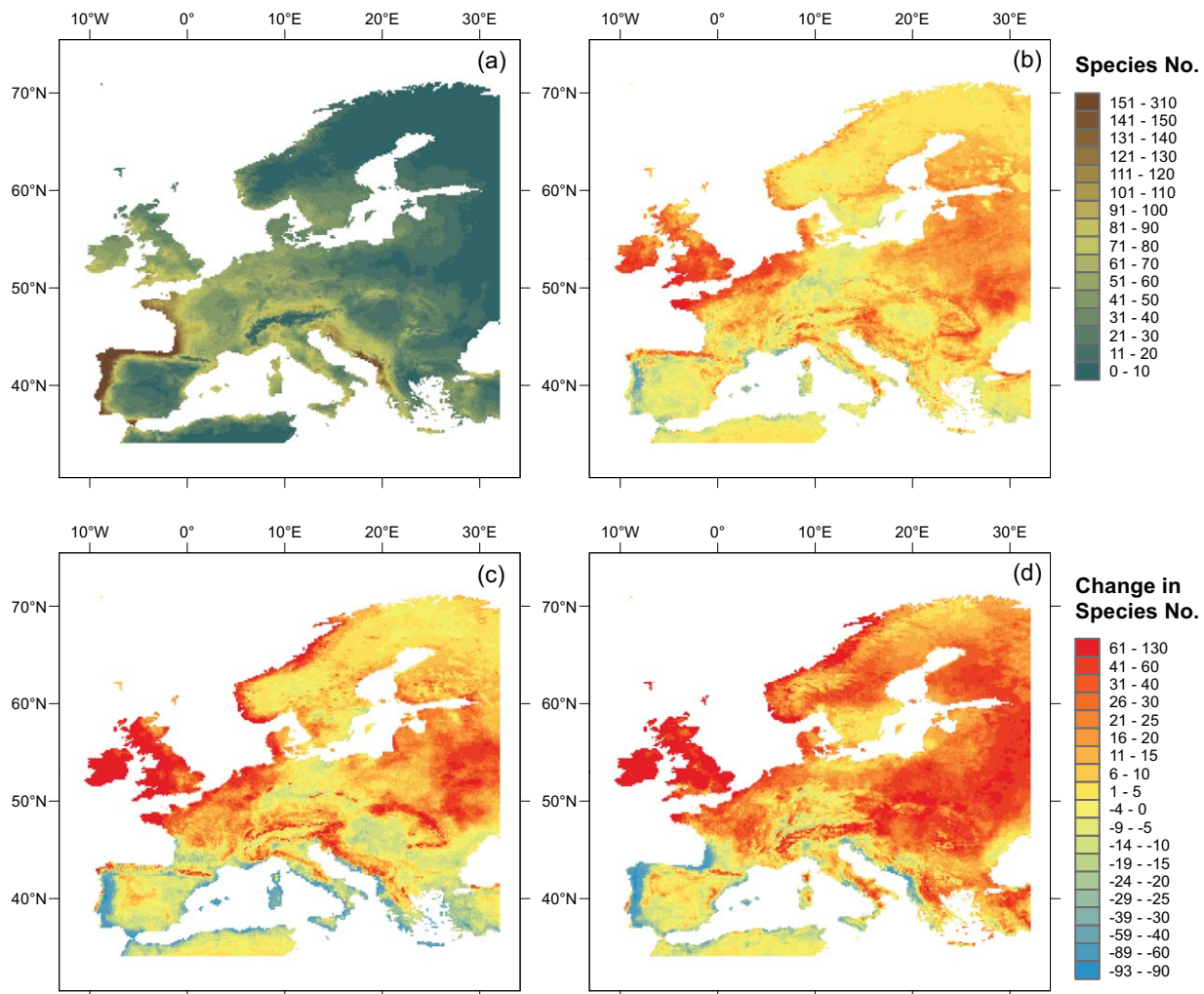


Figure 1 Projected climatic suitability for 783 ornamental species currently not naturalized in but somewhere outside of Europe in $10' \times 10'$ grid cells. The figure shows the total numbers of species that are projected to encounter climatically suitable conditions per grid cell under current climate (a), and changes to these numbers under three different climate change scenarios (b–d).

and by 102% under RCP8.5 (i.e. more than doubling) (Fig. 4b–d).

Although part of the southern Atlantic and the Balkan coasts will lose potential invaders under climate warming (Fig. 1), they nevertheless remain among those areas climatically suitable to a particularly high proportion of the analysed ornamental plants. The increasing extent of climatic hotspot area is mainly driven by a gradual expansion to the north including most of the British Isles, parts of north-western continental Europe, southern Norway and the western Pannonian region (Fig. 4b–d). However, most of northern and eastern Europe still does not qualify as a climatic hotspot, even under the most severe climatic scenario, although the number of potential invaders increases considerably there (Fig. 1b–d).

Risk maps

Similar to the extent of climatic hotspots, the area of high naturalization risk is predicted to grow under climate

warming by 28% under RCP2.6, by 30% under RCP4.5 and by 68% under RCP8.5 (Fig. 4f–h). Weighting by land-cover, however, results in some important changes to the purely climatic hotspot patterns (Fig. 4e–h). High-risk areas tend to extend further eastwards into densely populated areas of central and eastern Europe under all climate scenarios. By contrast, most of the Balkan coastal regions as well as parts of the Spanish coast are climatic hotspots under all scenarios but do not qualify as high-risk areas. Finally, parts of north-western Europe (e.g. Ireland, Scotland) and the southern Scandinavian coast become climatic hotspots when climate warms, but still do not appear to be areas with high naturalization risk.

DISCUSSION

Our results demonstrate that there is a sizeable pool of species which: (1) are planted in European gardens, at least locally, and hence already exert a certain amount of

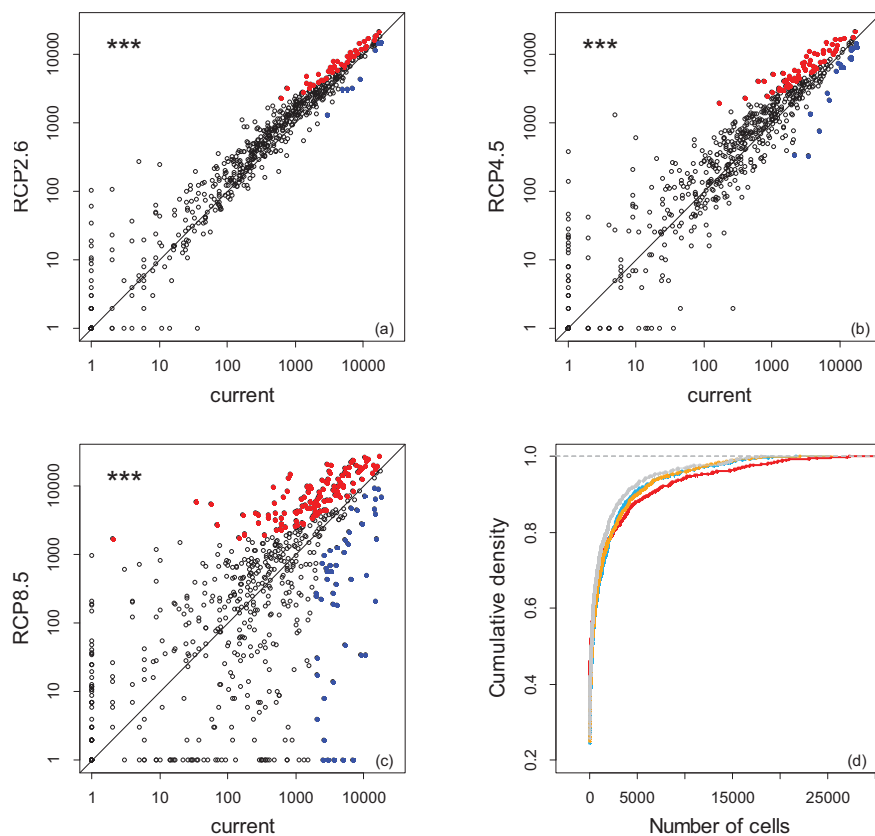


Figure 2 (a)–(c) Comparison of the number of cells climatically suitable for the 783 ornamental species under current climatic conditions and three different climate change scenarios (RCP2.6, RCP4.5, RCP8.5). Asterisks symbolize significant differences in the mean number of cells ($P < 0.001$). Blue and red points symbolize species that loose or gain > 1600 cells (c. 5% of the study area) in comparison with current climate conditions, respectively. (d) Cumulative density of the number of cells occupied by the species, i.e. the probability that a randomly selected species has a climatically suitable range $< x$ under current climatic conditions (grey), and under the three climatic scenarios (RCP2.6, light blue; RCP4.5, orange; RCP8.5, red). In (a)–(c) axes are log-scaled.

propagule pressure, (2) have proven their naturalization capacity in other parts of the world, and (3) find abundant suitable climatic space in Europe. The risk that at least some of these species will become naturalized in Europe in the future appears substantial, and it is likely that this risk will increase as climate change intensifies.

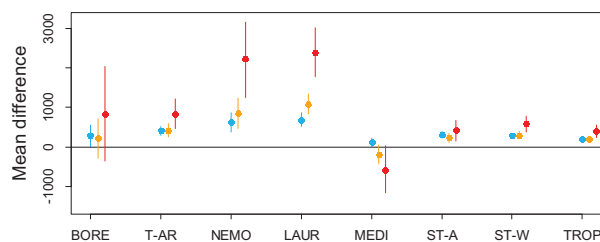


Figure 3 Mean difference in the number of cells climatically suitable to the 783 ornamental species under current climatic conditions and three different climate change scenarios (RCP2.6, RCP4.5, RCP8.5), separated by species zonobiome of origin. Points symbolize observed mean differences and lines 0.95 confidence intervals as derived from permutation tests. Key: blue, RCP2.6; orange, RCP4.5; red, RCP8.5; BORE, boreal; T-AR, temperate-arid; NEMO, nemoral (= temperate); LAUR, laurophyllous; MEDI, Mediterranean; ST-A, subtropical-arid; ST-W, subtropical seasonally dry; TROP, tropical.

Geographical distribution of current climatic hotspots

Climatically suitable areas for potential naturalization of garden plants are unequally distributed across Europe. Most parts of northern and eastern Europe are unsuitable for the vast majority of the analysed species under current climatic conditions, whereas hotspots are concentrated along the southern and western Atlantic shorelines and the eastern Adriatic coast. This geographical contrast suggests that not only temperature but a combination of temperature and precipitation regimes controls current patterns of climatic suitability for garden plants in Europe. The peculiarity of the Atlantic coastal areas, in particular, is a combination of relatively mild winters and humid summers keeping both frost and aridity stress low. These areas are hence likely to be within physiological tolerance limits of species from a wide array of different origins. By contrast, the Mediterranean region is warm enough in winter for nearly all selected species to be cultivated (Cullen *et al.*, 2011), but arid summers represent a climatic filter to naturalization. In line with this interpretation, the Balkan coastal area, which receives more precipitation than all other parts of the Mediterranean coast in Europe, is the only Mediterranean region that ranks among potential naturalization hotspots. In the eastern and northern parts of Europe, the climate is generally colder and/or more continental, with low winter temperatures, dry

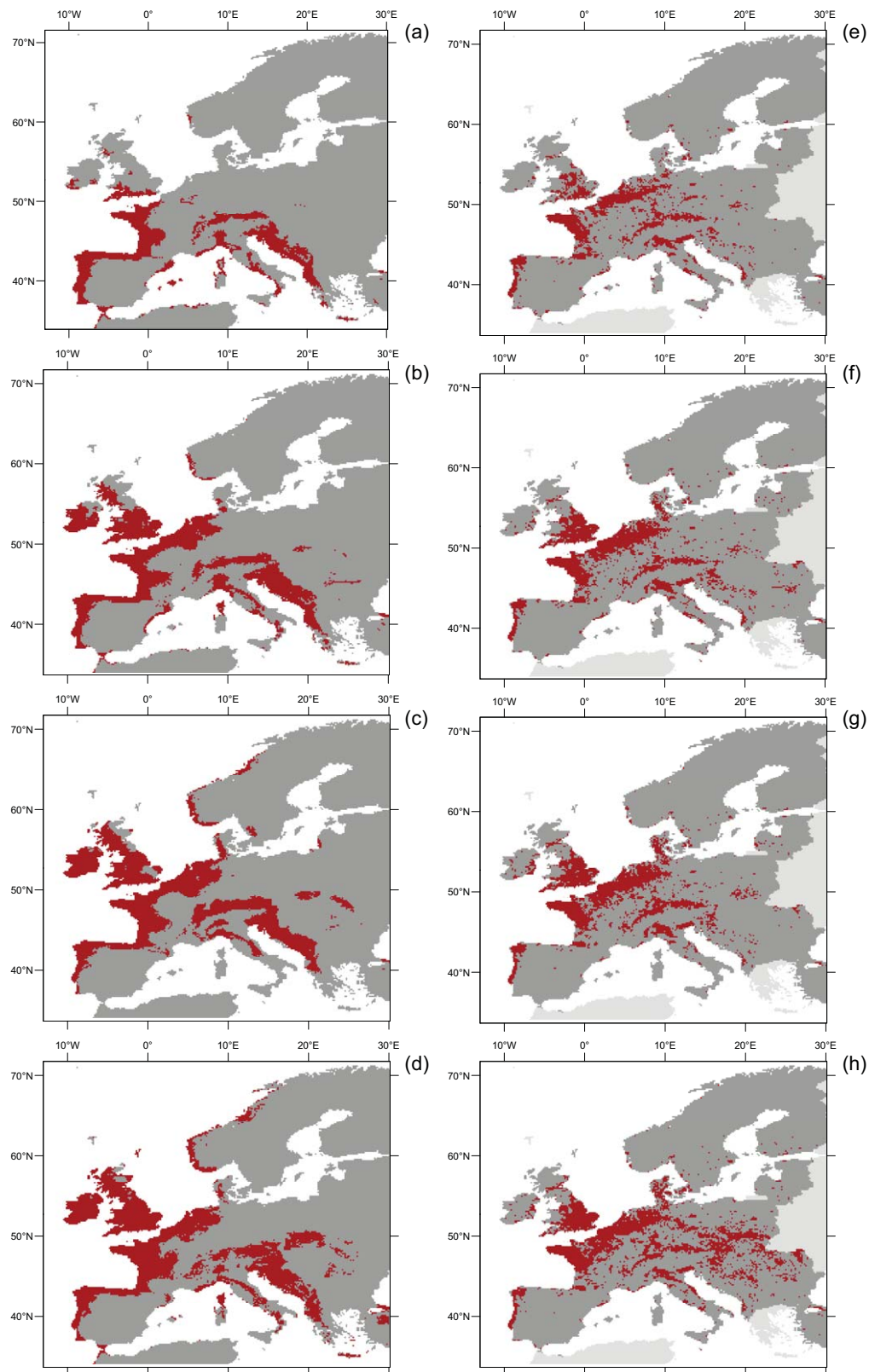


Figure 4 Geographical distribution of hotspots of potentially suitable climatic conditions for 783 ornamental species not yet naturalized in, but somewhere outside of Europe, under current climate (a) and three scenarios of climate warming: (b) mild scenario (RCP2.6), (c) intermediate scenario (RCP4.5) and (d) strong scenario (RCP8.5). (e)–(h) Maps of high naturalization risk calculated from combining climatic suitability under these four different assumptions of climatic conditions with the estimated area available for ornamental plant cultivation.

summers or a combination of both. These conditions are obviously hostile to the naturalization of most species from the current pool of European garden plants.

Effects of climate change

Release from climatic restrictions has been identified as a major potential driver of rising invasion risk under climate warming (e.g. Walther *et al.*, 2009). Our results generally support this notion. The prevailing pattern detected is an increasing number of potential invaders, in particular of laurophyllous and nemoral origin, in more northern and eastern parts of Europe and a concurrent shift of potential naturalization hotspots. This predicted expansion of climatically suitable ranges is particularly worrisome in the case of ornamental plants because many of them are already cultivated far beyond conditions that would currently allow population establishment in the wild (Van der Veken *et al.*, 2008). The presence of species propagules in regions that become newly climatically suitable to them effectively lifts dispersal limitations, and may therefore allow the naturalization of garden plants to keep track with climate change more closely than is commonly assumed for native plants (e.g. Corlett & Westcott, 2013).

The mean increase of climatically suitable area, however, masks pronounced variation among species. For a sizeable minority of the study species, the potential range is predicted to shrink under climate change, and under the most pronounced scenario the number of species finding suitable climate in <1% of the European area (320 cells) is approximately the same as under current conditions (442 vs. 441 species). The reasons for climatic range loss are likely to differ among individual species, but the fact that 'losers' are particularly widespread among species of boreal and Mediterranean origin suggests that two factors may be of particular importance. First, species adapted to cool conditions might lose potential area because temperatures become too warm in most parts of Europe. Second, species that would currently find climatically suitable area in Mediterranean Europe may not be able to deal with the more arid conditions that are predicted for these regions (Mariotti *et al.*, 2008) while, simultaneously, winter temperature does not become warm enough to compensate for such loss by expansion to the more northern, temperate parts of Europe. In accordance with the latter assumption, the regions that are currently both warm and relatively moist but will become drier in the future, like the southern Atlantic coast and the Balkan coastal area, are (1) predicted to lose the highest numbers of potential invaders and (2) are geographically separated from the more northern areas that show highest increases in the number of potential invaders.

Combining climatic suitability and potential ornamental planting area

Urban and suburban areas usually function as centres of introduction and cultivation for ornamentals, and the

proportion of introduced species usually decreases dramatically along an urban–rural gradient (Kowarik, 1995; Niinemets & Peñuelas, 2008). Combining projections of climatic suitability with the proportional area of the respective land-use types hence pinpoints some densely populated and economically prosperous regions in Europe as potential naturalization hotspots despite a sub-optimal climate, e.g. Great Britain under current climatic conditions. By contrast, relatively large areas appear less threatened although they would be climatically suited to many garden plants, at least under a warmer climate, like most of the coastal Balkan Peninsula, Ireland or some southern parts of coastal Scandinavia.

The risk maps presented here assume, however, that current land-cover patterns in Europe remain unchanged. Whether and how these patterns will change depends on future European socio-economic policies (Spangenberg *et al.*, 2012). Interestingly, a recent study projecting invasion levels in Europe as dependent on land-use change scenarios for the 21st century revealed patterns that partly resemble those found in our study, particularly with respect to rising naturalization risk in north-western and northern Europe (Chytrý *et al.*, 2012). Taken together, these parts of Europe will hence offer both climatically more suitable conditions and land-use patterns more susceptible to alien plant establishment in the future. By contrast, in the easternmost parts of the continent rising climatic suitability to potential invaders might be attenuated by abandonment and loss of former agricultural land in these economically marginal areas (Chytrý *et al.*, 2012; Spangenberg *et al.*, 2012).

Caveats

The use of species distribution models to predict range shifts under changing climatic conditions has important limitations, mainly related to the disregard of biotic interactions (e.g. Wisz *et al.*, 2013), intraspecific variation in niche breadth (Valladares *et al.*, 2014), dispersal limitations (Svenning & Skov, 2007) and, particularly in an invasion context, possible niche shifts (Early & Sax, 2014). In the case of our study, biotic interactions may be of limited relevance because the spatial resolution of our predictions is far beyond the scale at which plants usually interact (Pearson & Dawson, 2003). Likewise, dispersal limitation is probably less relevant as we model potential ranges of species that are actively distributed by humans, and for which the frequency of long-distance dispersal events can be expected to rise sharply in the future with the growing importance of e-commerce in the ornamental plant trade (Lenda *et al.*, 2014; Humair *et al.*, 2015). However, not all the plants modelled here will be traded and cultivated with equal intensity, and even of those planted frequently, only a subset will escape into the wild (Dehnen-Schmutz *et al.*, 2007). We hence stress that the numbers of species predicted in our study should not be taken at face value but represent a measure of spatial and temporal variation of naturalization risk. On the other hand, we note that the pool of potential invaders among European garden plants

might be even larger than assumed here because species could become established or even invasive in Europe although they have not yet done so in other regions of the world. Finally, with respect to niche shifts, we took care to parameterize our models not only with data from the native ranges of the species but also from all those areas where they have already naturalized. While this strategy should characterize the climatic potential of species as accurately as possible, further changes to realized niches during their possible future establishment and invasion in Europe can of course not be completely excluded.

The reliability of species distribution models depends on the quality of the data used to fit them. GBIF combines the advantage of global coverage, and hence the possibility to fit niches of species comprehensively, with the disadvantage of the errors and biases implicit in such large databases (Meyer *et al.*, 2016). However, we do not think that these errors and biases affect our results qualitatively. First, we took care to handle taxonomic problems and spatial errors when extracting occurrence data. Second, the poor coverage of northern Asia, and Russia in particular, which is probably the most important geographical bias of GBIF in our context, has little impact on our results as the number of species native to Russia in our pool is low (38 species). In addition, the detected increase of the invasion level is especially pronounced for species from nemoral and laurophyllous zonobiomes, which are mostly situated in regions with especially high record densities. Third, although predictions for individual species might suffer from inaccuracies, the multispecies patterns predicted here are consistently interpretable in terms of geographical gradients of climatic harshness in Europe, and hence appear highly plausible.

Conclusions

One of the greatest uncertainties in assessing the invasion risk of ornamental plants comes from the difficulty of estimating the potential impacts of climate change (Dehnen-Schmutz, 2011). Despite pronounced species-specific differences, our results suggest that climate warming leads to an increase in currently cultivated garden plants able to naturalize in Europe as well as the area across which they may spread. Which species will eventually become invasive or have a negative environmental and/or economic impact cannot be inferred from our models. However, a larger number of naturalized species probably also implies a greater risk of impact if the ratio of naturalized and harmful species remains about constant (Jeschke & Strayer, 2005). In addition, the growing importance of trade in ornamental plants via the internet (Humair *et al.*, 2015) increasingly removes any limitations on the availability of particular plants for the individual customer and hence largely eliminates the dispersal barriers that control range responses of non-cultivated species to climate warming (Svenning & Sandel, 2013). As a corollary, raising awareness of the invasion problem among individuals and institutions involved in gardening, urban landscaping and the

horticultural trade appears even more important in the face of a warming climate.

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SUPPORTING INFORMATION

Additional supporting information may be found in the online version of this article at the publisher's web-site:

Appendix S1 Selected species and number of suitable cells under current and future climate.

Appendix S2 Detailed model selection and downscaling procedure.

Appendix S3 Selection and weighting of relevant CORINE land-cover classes for risk map assessment.

Appendix S4 Information on model performance.

Appendix S5 Naturalization risk maps calculated according to three different weighting schemes.

Appendix S6 Species predicted gain or loss of area under climate change.

BIOSKETCH

Iwona Dullinger is a doctoral student with research interests in global change biology, conservation biology and social ecology. Her research mainly focuses on modelling the impacts of climate and land-use change on species diversity.

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MANUSCRIPT 3


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Will climate change increase hybridization risk between potential plant invaders and their congeners in Europe?

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Abstract

Aim: Interspecific hybridization can promote invasiveness of alien species. In many regions of the world, public and domestic gardens contain a huge pool of non-native plants. Climate change may relax constraints on their naturalization and hence facilitate hybridization with related species in the resident flora. Here, we evaluate this possible increase in hybridization risk by predicting changes in the overlap of climatically suitable ranges between a set of garden plants and their congeners in the resident flora.

Location: Europe.

Methods: From the pool of alien garden plants, we selected those which (1) are not naturalized in Europe, but established outside their native range elsewhere in the world; (2) belong to a genus where interspecific hybridization has been previously reported; and (3) have congeners in the native and naturalized flora of Europe. For the resulting set of 34 alien ornamentals as well as for 173 of their European congeners, we fitted species distribution models and projected suitable ranges under the current climate and three future climate scenarios. Changes in range overlap between garden plants and congeners were then assessed by means of the true skill statistic.

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Results: Projections suggest that under a warming climate, suitable ranges of garden plants will increase, on average, while those of their congeners will remain constant or shrink, at least under the more severe climate scenarios. The mean overlap in ranges among congeners of the two groups will decrease. Variation among genera is pronounced; however, and for some congeners, range overlap is predicted to increase significantly.

Main conclusions: Averaged across all modelled species, our results do not indicate that hybrids between potential future invaders and resident species will emerge more frequently in Europe when climate warms. These average trends do not preclude, however, that hybridization risk may considerably increase in particular genera.

KEYWORDS

alien ornamental plants, climate change, interspecific hybridization, invasion biology, range overlap, species distribution models

1 | INTRODUCTION

Biological invasions are an important component of global environmental change and may have severe ecological as well as economic impacts (Bellard, Cassey, & Blackburn, 2016; Vilà et al., 2011). Owing to intensified trade and traffic, the global redistribution of species and their subsequent establishment outside their native range (=their naturalization) have considerably increased during the recent decades and are likely to further increase in the future (Seebens et al., 2015). Pro-active management of such invasions is, however, hampered by the difficulty of predicting which species may become invasive and where. Such predictions are difficult because of the complex causes of invasions, which include biological traits of the invading species, biotic and abiotic characteristics of the recipient environment, and historical contingencies (Catford, Jansson, & Nilsson, 2009; Richardson & Pyšek, 2006). There are, however, a number of factors known to facilitate invasions such as early reproduction, rapid growth rate, efficient long-distance dispersal or specific trait profiles which are complementary to those of the resident biota (Buhk & Thielsch, 2015; Carboni et al., 2016; van Kleunen, Weber, & Fischer, 2010; Küster, Kühn, Bruehlheide, & Klotz, 2008; Pyšek et al., 2015).

Apart from these factors, interspecific hybridization has been assumed to foster invasions since a seminal paper of Ellstrand and Schierenbeck (2000). Indeed, there are prominent examples of highly invasive hybrids. For instance, several species of the genus *Tamarix* have been introduced to North America during the 19th century. Although all of these species have escaped cultivation, by far the most successful and widespread invader is the hybrid between *T. ramosissima* × *T. chinensis* (Gaskin & Kazmer, 2009; Gaskin & Schaal, 2002). The same Eurasian *T. ramosissima* has recently started to hybridize with native *T. usneoides* in South Africa (Mayonde, Cron, Gaskin, & Byrne, 2015). Other examples of genera that have produced successful invasive hybrids include *Rhododendron* (Milne & Abbott, 2000), *Spartina* (Thompson, 1991), *Senecio* (Abbott et al., 2009) and

Helianthemum (Rieseberg et al., 2007). More generally, the idea that interspecific hybrids may be especially successful invaders has been corroborated by a recent meta-analysis (Hovick & Whitney, 2014). The possible reasons for hybrid success include increased phenotypic or genotypic variability, phenotypic novelty arising from transgressive segregation or adaptive introgression, and heterosis effects (Prentis, Wilson, Dormontt, Richardson, & Lowe, 2008). Heterosis effects may be maintained especially when hybridization is accompanied by allopolyploidization and/or a shift to apomictic reproduction, which sustain heterozygosity.

As species are transported around the world with increasing intensity, barriers to gene flow between once geographically separated species are reduced and new hybrids between introduced and resident species will probably emerge more frequently (Thomas, 2013). For the British Isles, a recent overview has already demonstrated a rise in the number of hybrids during the last few decades (Stace, Preston, & Pearman, 2015). Apart from the risk that the new hybrids include particularly successful future invaders, rising hybridization rates also raise conservation concerns (Bohling, 2016). In particular, genetic introgression and outbreeding depression may severely threaten native species (Todesco et al., 2016), especially those that are rare and only exist in small populations (Bleeker, Schmitz, & Ristow, 2007).

Disregarding deliberate crossings (e.g., for horticultural reasons), the risk of hybridization between introduced and resident species will depend on the introduced species' ability to naturalize, that is to establish self-sustaining populations in the wild, because naturalization intensifies the spatial contact of the newcomers with their potential hybridization partners in the regional flora and hence increases mating opportunities. The likelihood of naturalization of an introduced species is mainly determined by propagule pressure (Simberloff, 2009) and the suitability of abiotic and biotic conditions (Pyšek et al., 2012; Shea & Chesson, 2002). Among the abiotic factors, climatic suitability has been repeatedly shown to play a

prominent role (e.g., Feng et al., 2016; Hayes & Barry, 2007; Thuiller et al., 2005). As a corollary, predicted climate change is also likely to alter the naturalization odds of introduced alien species and thus the likelihood that they hybridize with resident species (e.g., Bellard et al., 2013).

The alien flora of a region consists, first, of plant species that have already become naturalized or invasive. In addition, there is an often much larger group of alien species that have been introduced to a region and are grown there but have not escaped from cultivation yet. The latter group of species forms a massive pool of potential future additions to the regional wild flora. In many regions, this pool is dominated by non-native plants used for public and domestic gardening (Hulme et al., 2008; Niinemets & Penuelas, 2008; Pergl et al., 2016). In Europe, for example, more than 16,000 species from more than 200 families are currently in cultivation for ornamental purposes, with many of them being alien to Europe (Cullen, Knees, & Cubey, 2011). Some of these non-native garden plant species have already become naturalized or invasive elsewhere in the world (van Kleunen et al., 2015) and can hence be considered particularly likely to do so in Europe too (Williamson, 1999).

In a recent paper, Dullinger et al. (2016) showed that this latter group of “alien garden plants naturalized elsewhere” will benefit from a changing climate in Europe in as much as the area climatically suitable to them will increase. Given that climatic suitability is an important prerequisite to alien species’ naturalization and that naturalization facilitates hybridization of introduced and resident species, the risk that new hybrids emerge may thus also be expected to increase in the future. The newly establishing garden plants may thereby hybridize with resident (i.e., native and already naturalized or even invasive) species (e.g., Ayres, Smith, Zaremba, Klohr, & Strong, 2004). However, a climate-driven modification of regional hybridization risk does not only depend on the naturalization odds of garden plants, but also on changes in climatically suitable ranges of their potential hybridization partners (Dehnen-Schmutz, 2011). In other words, the changing spatial overlap in areas climatically suitable for alien garden plants and for their potential resident hybridization partners in the wild (both native and naturalized) flora will determine possible changes in the risk of hybridization between these two groups.

Here, we evaluated whether climate change may lead to an increase in this spatial overlap. We studied a group of 783 alien ornamental plants not yet naturalized in Europe, but established outside their native range elsewhere in the world, as identified in Dullinger et al. (2016). From this group of 783 species, we first selected all those belonging to genera with hybridization documented in the literature. We then fitted species distribution models for this subset of non-native ornamentals as well as for all their congeners in the native and naturalized European flora. We restricted our analysis to congeners because hybridization risk is strongly linked to genetic distance (Mallet, 2005), and intergeneric hybrids are rare (Whitney, Ahern, Campbell, Albert, & King, 2010). Finally, we assessed to what extent the range matching between the selected garden plants and their congeners will increase under three different climate change scenarios.

2 | METHODS

2.1 | Species selection

Our initial pool of study species was the same as used by Dullinger et al. (2016). These authors aligned the European Garden Flora (EGF; Cullen et al., 2011), the most comprehensive encyclopaedia of ornamental plants in Europe, with the Global Naturalised Alien Flora (GloNAF; van Kleunen et al., 2015; <https://glonaf.org/>), a global database of naturalized alien plant species. They thereby identified non-native ornamental plants cultivated in Europe which have naturalized somewhere outside of Europe, but not yet in Europe. For species distribution modelling (SDM) purposes, this list was then reduced to those 783 species with more than 50 occurrences found in a search of the Global Biodiversity Information Facility (GBIF, <http://www.gbif.org/>) database.

Here, we used a systematic web-based literature search to further narrow this group of candidate species to those particularly relevant in the context of both invasion and hybridization. We used all possible combinations of the following keywords in the Web-of-Science (<http://apps.webofknowledge.com>): #hybridization, #hybridisation, #invasion, #alien, #invasive species, #plant. The records were subsequently limited to the following categories: agriculture, biodiversity, conservation, ecology, environmental sciences, evolutionary biology and reproductive biology. We screened the abstracts of the 1,220 papers found and finally identified 66 plant genera that fulfil the following criteria: (1) interspecific hybridization has been documented and (2) they contain invasive species (even if these are not identical with the hybrids or if only intraspecific hybrids have so far been reported to be invasive, for example in *Pyrus* (Hardiman & Culley, 2010)). Twenty-three of these genera were represented by at least one species in the list of Dullinger et al. (2016), of which 18 were also represented by at least one species (native and naturalized) in the flora of Europe (Tutin et al., 1964–1980). From these, we discarded the genera *Rosa* and *Rubus* because of taxonomic difficulties with a large number of apomictic species. As a result of these consecutive filtering steps, we ended up with 16 genera. These 16 genera contain 34 alien plants currently cultivated in Europe with the potential to escape into the wild (indicated by their naturalization in other continents) and at least one congeneric species in the native and naturalized flora of Europe which shares the same life form (assuming that only mating partners of the same life form are likely to produce viable hybrid offspring; see Tables S1, S3, S6). Most of these species are planted for ornamental purposes only, but some, like *Chenopodium quinoa* or several *Eucalyptus* spp., are also of commercial interest beyond horticulture. After a final screening in GBIF for those species with more than 50 occurrence records (see Table S2), the group of congeneric species within Europe contained 133 native and 40 alien naturalized spp (see Table S6).

2.2 | Species distribution data and climatic maps

Data on the world-wide distribution of the 34 alien garden plants and their 173 native and naturalized congeners were taken from GBIF.

All species lists were taxonomically harmonized using The Plant List (<http://www.theplantlist.org>). Multiple occurrences within 10' × 10' grid cells and clearly erroneous records, that is those in water bodies, were removed. We did not limit records to those from the native range because species are known to partly expand their realized climatic niches in the naturalization range (Dellinger et al., 2016; Early & Sax, 2014; Petitpierre et al., 2012).

For characterizing the means and annual variability of the current temperature and precipitation patterns, we used six bioclimatic variables (climatic data averaged for the baseline period 1950–2000) provided by WorldClim (Hijmans, Cameron, Parra, Jones, & Jarvis, 2005): BIO4—Temperature Seasonality, BIO5—Max Temperature of Warmest Month, BIO6—Min Temperature of Coldest Month, BIO16—Precipitation of Wettest Quarter, BIO17—Precipitation of Driest Quarter, BIO18—Precipitation of Warmest Quarter. All these variables are known to potentially influence species distributions (Root et al., 2003). All climatic variables were provided by WORLDCLIM at a spatial resolution of 10 min.

Possible future climates in Europe were represented by three emission scenarios of the IPCC5-scenario family: the milder RCP2.6, the medium RCP4.5 and the severe RCP8.5 (IPCC, 2013). The respective monthly temperature and precipitation time series, already regionalized for Europe, were taken from the Cordex portal (<http://cordexesg.dmi.dk/esgf-web-fe/live>) and used to recalculate 10' resolution maps of the above six bioclimatic variables for possible future climates of the 21st century. A 50-year average of the period 2050–2100 was then used as the climate of the future in model projections (see below).

2.3 | Species distribution models

We used the BIOMOD2 platform (Thuiller, Lafourcade, Engler, & Araujo, 2009) in R (R Core Team, 2015) to quantify species' climatic niches and subsequently project current and future spatial distributions. The following modelling algorithms were used: generalized linear model (GLM), general additive model (GAM), boosted regression tree (BRT) and random forest (RF). For applying these species distribution models (SDMs) with presence-only data as provided by GBIF, we generated "pseudo-absences" following recommendations of Barbet-Massin, Jiguet, Albert, and Thuiller (2012): for regression technique models (GLM and GAM), we used 10,000 randomly distributed absences, and for machine-learning technique models (BRT and RF), we used a number of pseudo-absences equal to the number of occurrences found in GBIF and selected outside a radius of 200 km around these occurrences. In the latter case, pseudo-absence generation, and hence model calibration, was repeated 10 times per species to ensure that selected pseudo-absences did not bias the final predictions. For all models, the weighted sum of presences equalled the weighted sum of pseudo-absences. The predictive performance of the models was evaluated by means of the true skill statistic (TSS; Allouche, Tsoar, & Kadmon, 2006) based on a repeated (three times) split-sampling approach in which models were calibrated with 80% of the data and evaluated over the remaining 20%. Evaluated models were then used for two different average projections of the spatial distribution of

each of the 34 garden plants and their 173 native and naturalized congeners under current climatic conditions and the three climate change scenarios: one comprised the two regression-based techniques and one comprised the two machine-learning techniques. The probabilistic output of the two ensemble models was aggregated to a weighted mean, with weights determined by their respective TSS scores. Similarly, binary outputs of each of the two ensemble projections were generated based on a threshold that maximizes the TSS score (Liu, Berry, Dawson, & Pearson, 2005; Liu, White, & Newell, 2013) and then aggregated to a conservative consensus map; that is, 10' resolution cells were only classed as climatically suitable to a species if both ensemble models agreed on the potential presence of the species in the cell.

2.4 | Overlap of climatically suitable ranges

Geographic overlap between the climatically suitable ranges of the 34 alien garden plants and their 173 congeners under current and future climatic conditions was quantified by calculating the TSS from binary projections. Further, range overlap was quantified by the total number of overlapping grid cells, again based on binary projections. Both metrics were calculated for each possible species pair; that is, each of the 34 garden plants was combined with any of its congeners. Overlap metrics were subsequently averaged per species of garden plant (i.e., the average range overlap of each garden plant species and all its congeners in the wild flora was computed), separately for each climate change scenario. These average overlaps were then compared among the current climate and each climate change scenario using linear mixed-effects models (LMMs). Each LMM used the 34 ratios of current-to-future climatic range overlaps as the response, which was regressed against a fixed intercept, that is we tested whether the mean of the logarithm of these ratios was significantly larger or smaller than 0. A random intercept for genus was estimated to account for the fact that some genera were represented by more than one species of garden plant.

All analyses were carried out in R (R Core Team, 2015) mainly using the packages RASTER (Hijmans & van Etten, 2012) for handling of SDM gridded outputs, PRESENCEABSENCE (Freeman & Moisen, 2008) for calculating TSS and evaluation metrics and NLME (Pinheiro, Bates, DebRoy, Sarkar, & Team, 2015) for LMMs.

3 | RESULTS

3.1 | Geographic overlap of suitable ranges

Species distribution models for both the 34 alien garden plants and their 173 congeners in the native and naturalized European flora produced accurate projections in most cases (see Table S6).

True skill statistic scores suggest that the mean geographical overlap between the climatically suitable ranges of the 34 garden plants and their congeners will decrease under a warming climate (Figure 1a): the overlap is lowest under the strongest scenario (RCP8.5) and also significantly different from current climatic conditions under the

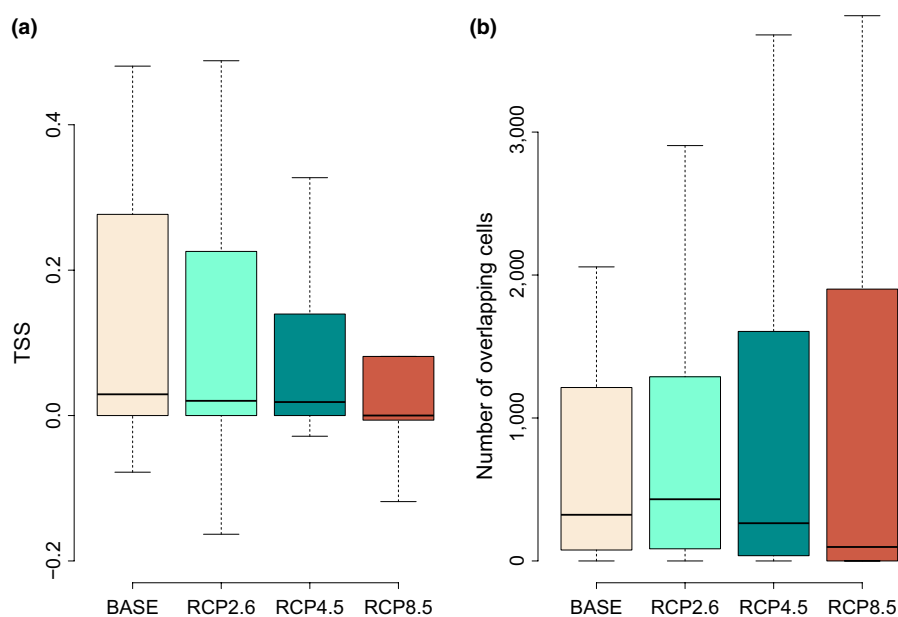


FIGURE 1 Mean overlap in areas climatically suitable to 34 alien garden plants and their congeners in the native and naturalized flora of Europe. Overlap was quantified by the true skill statistic-TSS (a), or the number of overlapping cells (b), and calculated for current climate (BASE) and under three scenarios of climate change (RCP2.6, RCP4.5, RCP8.5) for the second half of the 21st century (2050–2100)

mild and intermediate scenarios (RCP2.6 and RCP4.5; see Table S4). When overlap is measured as the number of $10' \times 10'$ cells that are climatically suitable to both the garden plants and their congeners (i.e., the absolute size of their overlapping range, see Table S5), the results suggest that a warmer climate will not change the size of overlapping ranges in a statistically significant way in any of the scenarios (Figure 1b, see Table S4).

Looking at climatically suitable ranges of the 34 garden plant species and their 173 congeners separately indicates that these results are partly driven by opposite effects of climate change on the two species groups: while average range size (=number of suitable cells) is projected to increase for the garden plants (statistically significantly only for scenario RCP8.5, see Table S4), it will remain constant or even decrease for their congeners in the wild European flora, at least under

the more severe scenarios (RCP4.5 and RCP8.5, Figure 2 and see Table S4). These opposite trends apparently result in no net change in overlap or in a slight reduction depending on scenario and overlap measure used, but never in a significant increase in overlap.

These average trends mask strong differences among genera. Figure 3 demonstrates that the number of cells climatically suitable to both the 34 garden plants and their European congeners can either strongly decrease or increase under each of the future climate scenarios, and variation among individual species pairs (i.e., a particular garden plant species with all its individual congener species) is even more pronounced. In particular, under each of the scenarios, there are a number of genera for which spatial overlap of suitable ranges between non-native ornamental plants and their European congeners will increase markedly. This is especially true for the genera *Solidago*, *Fraxinus*, *Lonicera* and *Prunus*.

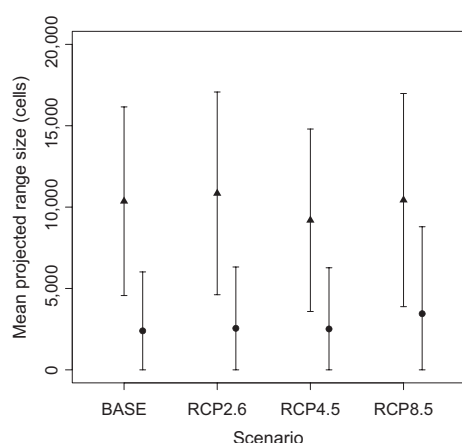


FIGURE 2 Mean projected range size of 34 alien garden plants (circles) and of their 173 congeners in the native and naturalized flora of Europe (triangles) under current climate (BASE) and under three different scenarios of climate change (RCP2.6, RCP4.5, RCP8.5) for the second half of the 21st century (2050–2100). The error bars indicate the standard deviation

4 | DISCUSSION

Taken together, our results do not support the expectation that the area suitable to both the group of potential future invaders among European garden plants and their congeners in the resident flora of the continent will increase under a changing climate. Potential range overlap between these two groups of species will rather decrease under all warming scenarios. This is partly due to opposing trends in the size of climatically suitable ranges among the two groups: while potential invaders on average expand their suitable ranges, those of resident congeners remain constant or shrink, at least under moderate and severe warming. However, there is pronounced variation among the different species pairs and for some of them the predicted increase in range overlap is significant, suggesting that the risk of hybridization between them will also increase.

Climate change has already allowed many alien species to expand their non-native ranges (e.g., IPCC, 2014). For ornamental plants, the

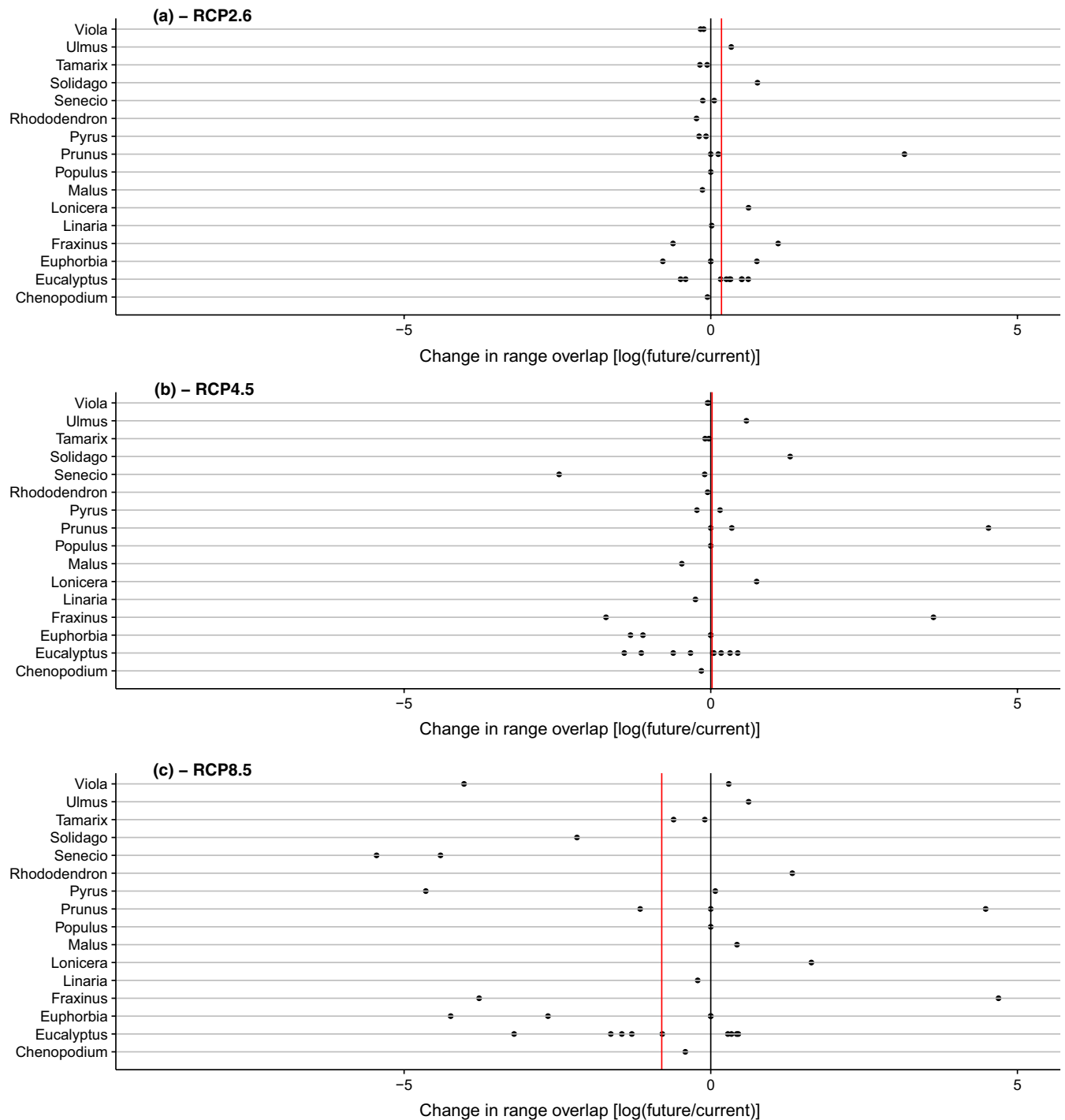


FIGURE 3 Change in overlap of areas climatically suitable to 34 alien garden plants and their 173 congeners in the native and naturalized flora of Europe. Overlap in areas is measured by the log of the ratio of the number of $10 \times 10'$ cells suitable to both species in a possible species pair. Each point represents the average change in overlap between one of the 34 garden plants and all its congeners under the respective climate scenario (some points represent more than one pair because of identical values). Values <0 represent a decrease, values >0 an increase, values $=0$ no change in overlap. The three panels refer to climate change scenarios RCP2.6 (a), RCP4.5 (b) and RCP8.5 (c). The red line represents the mean over all pairs

main reason for this trend is probably their widespread commercial use beyond climatic conditions they would tolerate in the wild, which gives them a head start when the climate warms (Van der Veken, Hermy, Vellend, Knapen, & Verheyen, 2008). Predictions of increasing suitable range sizes of ornamental plants in a warming Europe likely

have similar underlying reasons. Many ornamentals currently cultivated on the continent come from warm(er) regions and hence tend to expand towards north-eastern and north-western Europe, in particular, if climatic constraints in these regions are relaxed (cf. Bellard et al., 2013; Dullinger et al., 2016). The 34 non-native ornamental

plants used in this study are also mostly native to warm regions and hence their potentially suitable ranges in Europe tend to increase, on average, despite pronounced idiosyncratic differences. Although the pool of their European congeners contains many warm-adapted species too (e.g., most species from the genera *Euphorbia* and *Tamarix*), it also includes a considerable number of montane or even alpine species (e.g., from the genera *Linaria*, *Rhododendron*, *Senecio* and *Viola*). For montane species, climatically suitable ranges are particularly likely to shrink under climate warming (Engler et al., 2011; Thuiller et al., 2014). The share of montane species is thus probably a factor restricting range increases of congeners in the more severe climate scenarios.

We emphasize that our estimate of changing range overlaps does not include a temporal dimension. Real changes in overlap of species distribution over the 21st century may actually deviate from those projected here. On the one hand, wild populations of species (both native and naturalized) will likely lag behind the changing climate due to dispersal and migration constraints (e.g., Corlett & Westcott, 2013; Dullinger et al., 2015). These constraints are less relevant or even irrelevant for ornamental plants in horticultural trade. Actually, garden plants may even “overtake” climate change when regional demand of gardeners anticipates future climatic alterations (Bradley et al., 2012). On the other hand, remnant populations of species in the wild may still occupy an area long after the average climate has become unsuitable to them (Eriksson, 2000). Actual range overlap over the next decades will hence not only be a function of changes in suitable ranges, but will be co-determined by the behaviour of gardeners and by migration lags and extinction debts of wild populations (Dullinger et al., 2012). Thus, we may expect that our SDM-based projections will underestimate real overlap near the wild species' trailing edges (because of delayed extinctions), but overestimate it near the wild species' leading edges (because of lagged migration).

An average decrease in range overlap among all the species pairs tested here does not necessarily imply a general decrease in hybridization risk from invasive plants in Europe. First, we deliberately restricted our approach to hybridization among potential future invaders and resident species but did not consider the possible emergence of hybrids within the resident (i.e., native and already naturalized or even invasive) species. Among the latter, several hybrids come from genera well-known to hybridize such as *Fallopia* (Parepa, Fischer, Krebs, & Bossdorf, 2014) or *Epilobium* (Gregor et al., 2013). For an exhaustive evaluation of climate-driven changes in hybridization risk of non-native plants, these species would have to be included into the models. Second, the probability of hybridization risk will likely vary widely among the species pairs included in this study. Successful establishment of allopolyploid hybrids, for example, depends on plant traits (Mallet, 2007). In addition, the genetic distance between species certainly differs a lot among the pairs studied and hence also the likelihood that reproductive barriers break down (Mallet, 2005). A more precise evaluation of hybridization risk under climate warming would therefore have to weight changing range overlaps by the likelihood that particular species pairs hybridize at all—and, in an additional step, by the probability that a particularly successful invader emerges from

such hybridization (e.g., Abbott et al., 2009; Hovick & Whitney, 2014). Such weighting might significantly modify expected changes in hybridization as individual species pairs with increasing range overlap are to be found in almost all genera. Although data for reliable estimation of these weights are lacking, we emphasize that among the genera with increasing average range overlaps in at least some scenarios, species in *Solidago* and *Rhododendron* have already produced invasive hybrids in Europe (Abbott et al., 2009; Erfmeier, Tsaliki, Ross, & Bruehlheide, 2011; Karpaviciene & Radusiene, 2016) and may hence be particularly likely to do so in the future again. In addition, among the genera which were both identified to have produced invasive hybrids in the meta-analysis of Hovick and Whitney (2014) and used in our study, three include species pairs with increasing average range overlaps in at least some climate change scenarios (*Rhododendron*, *Ulmus*, *Viola*) and only one solely contains pairs with decreasing average overlap (*Tamarix*).

Although we consider the change in suitable range overlap to be a sensible indicator of changing hybridization risk, the emergence of hybrids does not necessarily depend on the contact of the species in the wild. Some of the native or already naturalized congeners in our study are species that frequently occur at ruderal sites or even as garden weeds (e.g., *Euphorbia peplus*, *Senecio vulgaris*) and hence also potentially reproduce with plants cultivated in gardens or parks. For these species, changing hybridization risk might more realistically be estimated from how their future suitable ranges overlap with the possible area where potential hybridization partners among ornamental plants can be cultivated when climate warms. These areas are usually much larger than those suitable for establishment of wild populations (Van der Veken et al., 2008) and hence risk assessments based on the latter may actually be underestimates.

Apart from potentially fostering invasiveness, hybridization between alien and native plants may threaten native populations of rare species through outbreeding depression (Bleeker et al., 2007), gene swamping (Todesco et al., 2016) or pollen competition (Arceo-Gomez & Ashman, 2016). Among the genera included in this study, introgressive hybridization has been documented in several cases (e.g., *Tamarix* (Gaskin & Kazmer, 2009), *Rhododendron* (Stace et al., 2015), *Viola* (Stace et al., 2015)). Conversely, Bleeker et al. (2007) have identified 18 native species red-listed in Germany, which potentially suffer from outbreeding depression when hybridizing with more abundant aliens. Among the 13 genera these species belong to, six are also included in our study (*Euphorbia*, *Malus*, *Populus*, *Prunus*, *Solidago*, *Viola*) with two of them (*Solidago*, *Prunus*) tending towards increased range overlap with native congeners under a warming climate (these results are very similar when the climatic area of natives that is also suitable to their non-native congeners among garden plants is calculated as a measure of threat to the native plants, see Fig. S3). In addition, Bleeker et al. (2007) listed threatened native *Viola* spp. as sensitive to gene introgression from alien congeners. Similar evaluations for other European countries are largely lacking. However, across Europe, the congeners of our 34 potential future invaders include many regionally endangered or even globally rare species such as Mediterranean endemics in the genera *Linaria*, *Senecio* or *Viola*. Although the magnitude of threat to rare species from outbreeding depression and introgression with

hybridizing aliens is not well documented yet (Bohling, 2016), future escape and expansion of ornamental plants into the range of these endemics may actually put additional pressure on them, beyond the challenges they face under a warming climate. Most of these species are not included in our study as their distribution is not represented well enough in GBIF, but this issue certainly warrants further investigation.

Finally, as a last caveat, we note that the models this study is based on were fitted using data taken from GBIF. This source combines the advantage of a global coverage, and hence the possibility to fit niches of species comprehensively, with the disadvantage of the errors and biases implicit to this database (Meyer, Weigelt, & Kreft, 2016). Uncertainties in species distribution estimates and models resulting from these caveats have a clear geographical bias and are least pronounced in the well represented regions of Europe, North- and Central America, and Australia (Meyer et al., 2016). The majority of the ornamental plants and all congeners modelled here come from these areas, and we hence assume that data problems are of limited importance for them. Several of the ornamental plants are native to temperate Asia and Africa, however, and these regions have notoriously low data coverage. The most likely consequence of this low coverage is an underestimation of these species' niches and hence of their potential distribution in Europe as well as their overlap with native and already naturalized congeners. Such underestimation may have been reinforced by the restrictive rules of our consensus projections. As a result, range overlap estimates computed here are probably conservative. We do not, however, think that these data problems affect our main result, namely that the average potential range overlap between ornamental plants and congeners does not increase under a warming climate. This is because predicted trends for species of Asian and African origin are similar to those of the remaining species (see Fig. S1 and Fig. S2 respectively).

5 | CONCLUSION

Climate warming will potentially increase the area suitable for the naturalization of many non-native ornamental plants in Europe (Dullinger et al., 2016), but the mean geographical overlap of climatic ranges between the selection of ornamentals and their native and naturalized congeners modelled here is unlikely to increase in the future. Thus, the average risk that garden plants and their wild congeners in the European flora will hybridize does not appear to rise when climate warms. We emphasize, however, that suitable range overlaps do increase for many individual congener pairs and that the pair-specific likelihood of successful hybrid establishment is unknown. A decreasing average range overlap does not, therefore, preclude increasing invasion risk from hybrids between particular species pairs.

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AUTHOR CONTRIBUTIONS

S.D. and G.K. conceived the ideas; G.K. and A.G. analysed the data; I.D. and J.W. contributed to the analyses; G.K. and S.D. led the writing of the manuscript; all authors contributed to the discussion of ideas and revised the text.

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BIOSKETCH

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SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

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Effects of climate change and propagule pressure on ornamental plant invasions in Europe

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Biological invasions are of primary concern in environmental policy¹ but current mitigation strategies may be challenged by suspected accelerating effects of climate change on the spread of alien species^{2, 3, 4}. However, few studies have assessed the efficacy of invasive species policies under expected future climates so far. Here, we use coupled niche-demographic modelling to simulate the spread of 15 potentially invasive ornamental plants across Europe during the 21st century. We thereby assess the effects of predicted climate change and assumed restrictions of cultivation on simulated dynamics. We find that the ranges climatically suitable to these species increase, on average, under all tested climate scenarios. Nevertheless, the mean area simulated to be occupied by the end of the century is smaller than under a constant climate for two out of three scenarios because species colonize suitable areas with delay and part of the transiently established populations go extinct again. By contrast, reducing cultivation frequency has a strong, consistent, but non-linear effect on species spread across all climate change scenarios. Restrictions of use thus depend on high levels of compliance for being efficient. Such compliance provided, they can successfully reduce plant invasions to low levels, irrespective of how the climate develops.

The number of species establishing populations outside their native ranges has increased exponentially during the recent century, and this trend is unlikely to slow down in the future⁵. Some of these alien species become invasive, i.e. they manage to spread rapidly and maintain large populations over an extended area⁶. These invaders may threaten native biodiversity^{7, 8, 9} and can have negative effects on ecosystem services, the economy^{10, 11} and human health¹².

In the case of vascular plants, horticultural trade is known to be the most important pathway of alien species introductions¹³. Tens of thousands of introduced plant species are cultivated in private and public gardens and green spaces worldwide, and represent a seemingly inexhaustible pool of potential future invaders¹⁴. Climate change is expected to increase the likelihood that further species from this pool escape cultivation and eventually become invasive for two reasons. First, an emerging disequilibrium with climatic conditions¹⁵ may destabilise resident communities and decrease their biotic resistance to the establishment and spread of alien species^{4, 16}. Second, warmer temperatures may expand the area climatically suitable to escaping ornamentals, especially in temperate regions where many garden plants have been introduced from warmer native ranges^{17, 18}. As human cultivation represents an efficient dispersal pathway, areas becoming suitable under climate change may, moreover,

become more rapidly colonized by escaping ornamentals than by non-cultivated species which have to rely on natural dispersal means^{19, 20}. However, the implications that cultivation pattern and frequency may have on the possible spread of alien ornamentals under a changing climate has hardly been explicitly evaluated so far.

Assessing the relationship between cultivation intensity (i.e. the size and frequency of the populations introduced for horticultural purposes) and spread of alien ornamentals is also topical from the perspective of environmental policy¹. Import or sales bans, voluntary codes of conduct, and raising consumer awareness are the main instruments used to tackle invasions along the horticultural supply chain²¹. While none of these measures is sufficient to completely prevent cultivation of potentially invasive garden plants, each of them can significantly contribute to reducing cultivation intensity. Given the importance of propagule pressure for many invasions in general²², and the demonstrated correlation between cultivation intensity and the likelihood of ornamental escape²³ in particular, these measures may thus still be effective despite imperfect compliance. Evaluating how levels of compliance with cultivation restrictions relate to their efficacy for preventing invasions would hence provide important information for implementing regulations appropriately.

Here, we undertook a large-scale simulation experiment to explore the impact of climate change and cultivation restrictions on the spatio-temporal spread of 15 ornamental plant species (Supplementary Table S1) across Europe during the 21st century (until the year 2090). The 15 species include annuals and perennial herbs as well as graminoids which differ in a number of demographic and dispersal related traits as well as in habitat affinities, and originate from different parts of the world (Supplementary Table S2 and Table S3). They hence represent a broad spectrum of the non-woody European garden flora. All selected species are currently cultivated in Europe but have not escaped from cultivation on the continent so far. All of them have, however, established alien populations outside Europe and have hence already demonstrated their naturalization capacity²⁴. We moreover only selected species for which a previous study has demonstrated that the area climatically suitable to them in Europe will remain constant or increase under at least one scenario of climate change¹⁸.

We simulated the escape and spatial spread of these species by means of coupled niche-demographic modelling²⁵, with two different sets of dispersal parameters to cover uncertainty in and explore sensitivity to these parameters (called ‘high dispersal’ and ‘low dispersal’ sets henceforth, see Supplementary Table S2). Simulations were run under an assumed constant climate (‘baseline’) and three different scenarios of climate change (moderate RCP 2.6,

intermediate RCP 4.5, severe RCP 8.5; Supplementary Table S4). The simulation design involved cultivation of the 15 model plants in randomly selected gardens across climatically suitable parts of Europe at six levels of frequency (0.01% - 10% of gardens and green spaces, see Supplementary Methods for details). We assumed that these levels span a gradient from a situation where cultivation restrictions are in force and compliance is high (although not perfect) to an unregulated situation for popular ornamentals.

Our simulations (Supplementary Table S5) confirm that 12 of the 15 species will gain climatically suitable area by the end of the century under at least one scenario of climate change (9, 12, and 8 species under RCP 2.6, RCP 4.5 and RCP 8.5, respectively, see Supplementary Fig. S1). However, despite the average increase in suitable areas, the effect of a changing climate on the spatio-temporal spread of species is ambiguous (Fig. 1A, and Supplementary Fig. S2A). Although under the intermediate RCP 4.5 scenario the area occupied in 2090 is significantly larger than under the baseline climate (Table 1 and Supplementary Table S6), the opposite happens under the moderate and severe scenarios RCP 2.6 and 8.5, respectively. The reasons behind are, first, that species are unable to colonize all suitable area until the end of the century, even under a constant climate (compare the still strong increase in the number of occupied cells towards the end of the simulation period in Fig. S3); and, second, that suitable sites become spatially displaced under a changing climate: proportional loss rates of populations (from one decade to the next) are high early in the simulation period due to fluctuating occupancy of marginally (un)suitable sites and thus high demographic stochasticity, but stabilize or decrease after the first decades when an increasing stock of suitable sites has become occupied (Fig. 1B, Supplementary Fig. S2B). For moderate and severe scenarios RCP 2.6 and 8.5, however, loss rates increase again towards the end of the century, indicating that part of this stock of suitable sites has become unsuitable again, and populations go extinct (Fig. 1B, Supplementary Fig. S2B and Fig. S4). In the intermediate RCP 4.5 scenario, the effect of these extinctions is probably outweighed by the larger average increase in the area suitable to the species (Fig. 1A). In the other two scenarios, the net balance is negative, either because the amount of suitable area added is smaller (RCP 2.6) or because the spatial displacement of suitable areas is more pronounced (RCP 8.5, Supplementary Fig. S4).

The effect of cultivation frequency on the area occupied by naturalized populations of the 15 species in 2090 is non-linear: spread rates increase sharply between 0.01 % and 1% of cultivation frequency but this increase levels off subsequently (Fig. 2A and Supplementary

Fig. S5). However, as compared to climate change, the effect of cultivation frequency on species spread is much stronger. Together, the two factors explain approximately 62% and 64% of the variation in the area occupied by wild populations of the 15 species in simulations with high and low dispersal parameter values, respectively (Table 1 and Supplementary Table S6), but more than 90% of this explained variation is due to the level of cultivation frequency. The interaction between cultivation frequency and climate change scenarios is statistically significant (Table 1 and Supplementary Table S6), with restrictions of use being strongest under an intermediate climate change scenario (RCP4.5, cf. Fig. 2A). However, these interactive effects are weak and explain little additional variation in the simulation results (Table 1 and Supplementary Table S6). Across all species, the evaluated three orders of magnitude increase in cultivation frequency resulted in six orders of magnitude increase in the area simulated to be occupied by escaped populations in 2090, irrespective of the climate change scenario (Supplementary Fig. S3). This huge effect is mainly driven by three species (*Heliotropium arborescens*, *Pennisetum macrourum*, *Verbena rigida*), which profit disproportionately from higher cultivation intensity (Supplementary Fig. S6). For the other 12 species, the effect of a higher cultivation frequency is weaker, although still pronounced. As a consequence, in 2090, the maximum number of species simulated as established, exemplarily calculated for grid cells of 20 x 20 km², rises from 1 to 10 when increasing cultivation frequency from 0.01 to 10 % (Fig. 3).

In summary, our results suggest that the spread of potentially invasive garden plants in Europe will not necessarily be fostered by a changing climate, at least until the end of the 21st century, even if human cultivation relaxes dispersal limitations^{19, 20}. The discrepancy between larger suitable and smaller occupied areas in two of the three scenarios suggests, however, that an invasion debt²⁶ is accumulating which may be paid off when the climate stabilizes again. In addition, our results may underestimate climate change effects for two reasons. First, the simulations only accounted for changes of the area climatically suitable to the alien species and did not incorporate possible alterations of native vegetation under climate change¹⁵ which may increase the invasibility of resident communities^{4, 16}. Second, the changing climate may be associated with altered land use regimes²⁷ which may also foster invasibility, at least in parts of Europe²⁸. On the other hand, our selection of model species excluded those that will lose climatically suitable area in Europe under climate change. While those species are likely fewer in number than the ‘winners’¹⁸, such a biased selection must necessarily overestimate the average positive effect of a warming climate across all possible future invaders from the European garden flora.

While our simulations do not support the concern that climate change will greatly accelerate invasion of alien ornamentals in Europe, they underline the strong effects that restrictions of use can have on the spread of introduced garden plants. Their escape and spread can probably not be completely prevented, but it can be greatly reduced when cultivation intensity is kept at low levels. These effects are largely independent of future climate change scenarios. However, the non-linear dependence on cultivation intensity implies that only high rates of compliance with invasive species regulations will render restrictions fully effective. To achieve such compliance different policy instruments like risk assessments, legal regulations, industry codes of conduct and raising customer awareness will have to be combined²¹. Our results emphasize that ornamental plant invasions can be considerably reduced by an efficient integration of such measures, irrespective of how the climate develops.

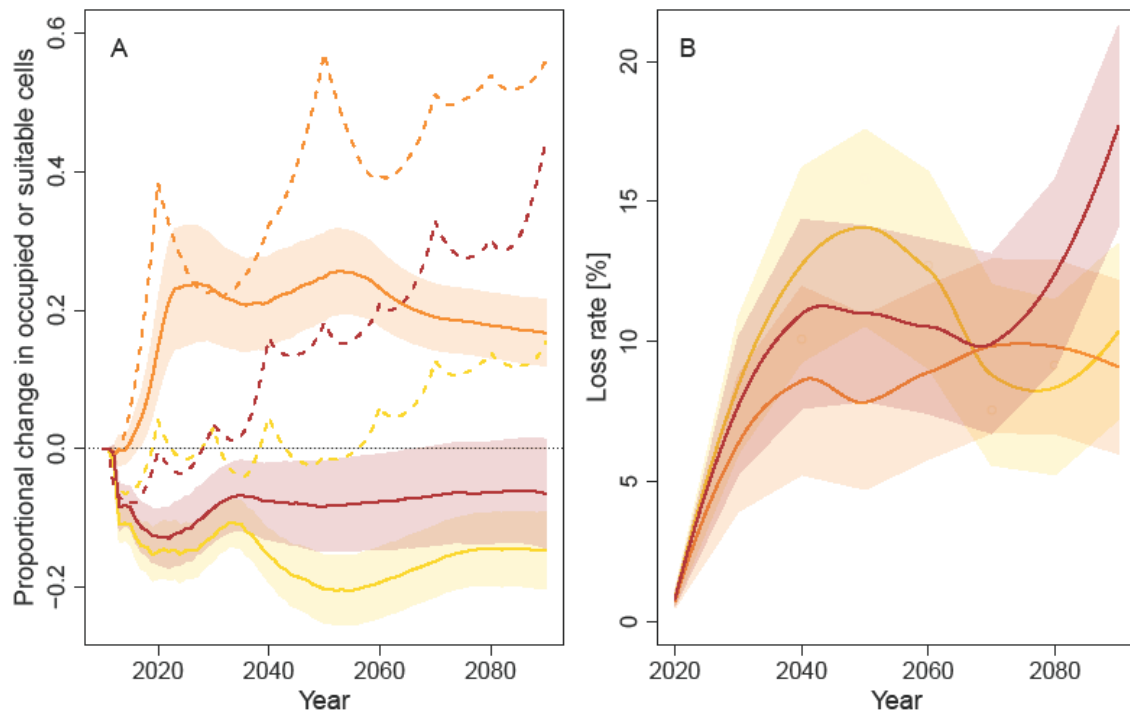


Fig. 1: Simulated future development of surface area occupied by 15 alien ornamental plants under climate change in Europe. Colours indicate different climate change scenarios: ‘yellow’ – moderate RCP2.6, ‘orange’ – intermediate RCP4.5, ‘red’ – severe RCP8.5; lines represent averages over all 15 species and shaded areas indicate standard errors. A: The proportional change in the number of cells occupied (solid lines) and climatically suitable (dashed lines). B: Percentage loss of occupied cells between two consecutive decades. The presented results are for simulations with cultivation frequency of 1 % and dispersal parameters set to ‘high’. Results with ‘low’ dispersal parameters are qualitatively similar (Supplementary Fig. S2).

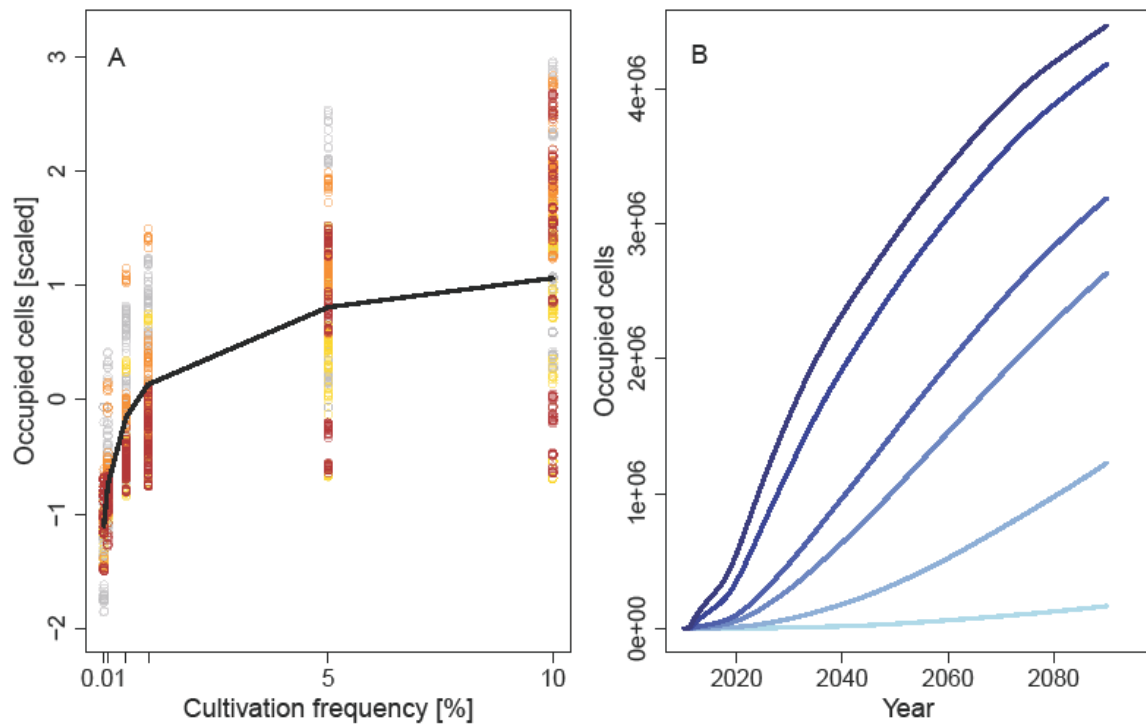


Fig. 2: Effect of cultivation frequency on the simulated spread of 15 alien ornamental plants in Europe. A: The average area occupied by the species at the end of the simulation period (year 2090), measured as number of 250 x 250 m² cells. Cell numbers have been scaled to a mean of 0 and a standard deviation of 1, separately for each species. Circles represent results of individual simulation runs, with colours indicating climate change scenarios (‘grey’ – Baseline, yellow’ – moderate RCP2.6, ‘orange’ – intermediate RCP4.5, ‘red’ – severe RCP8.5). B: The average number of cells occupied by the 15 species over the simulation period. Colours indicate different cultivation frequencies from 0.01 % (light blue) to 10% (dark blue). The results represent simulations under the RCP 4.5 scenario and dispersal parameters set to ‘high’. Results under low dispersal and other climate change scenarios are qualitatively similar (Supplementary Fig. S5).

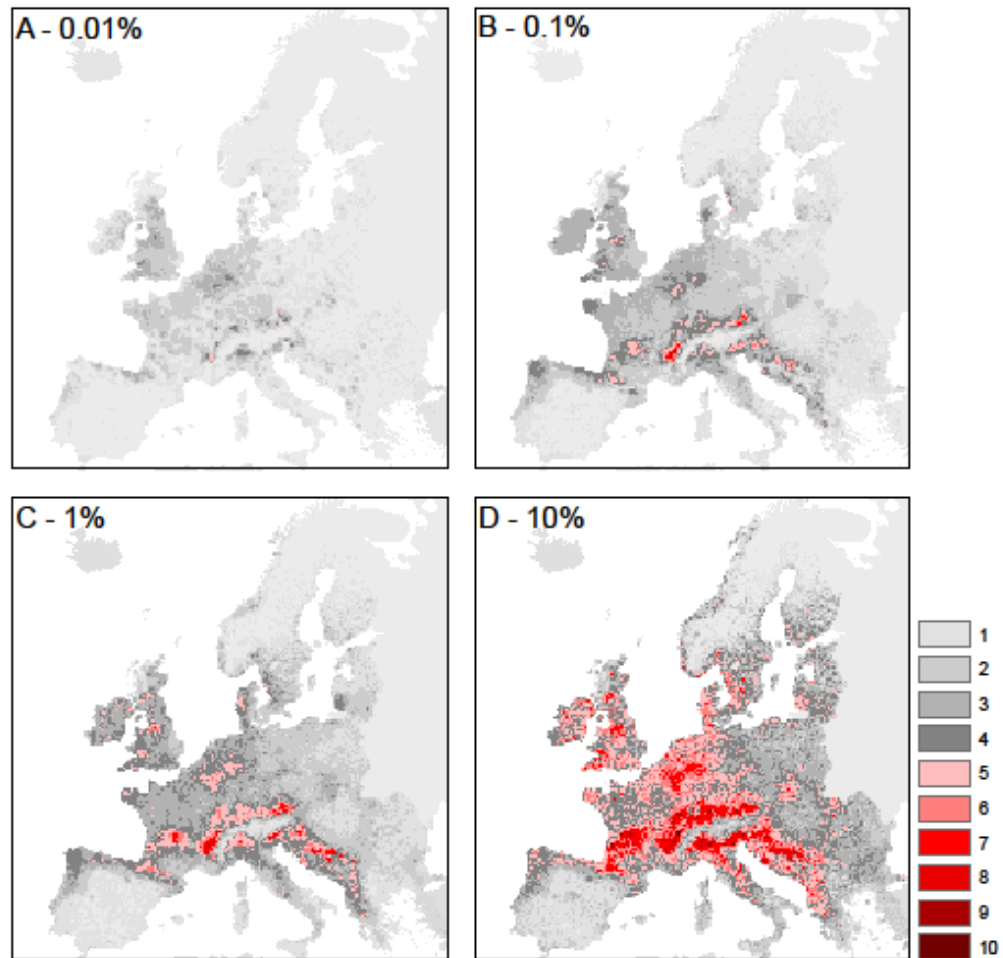


Fig. 3: Number of naturalized alien ornamental species simulated to co-occur in 2090. Numbers have been calculated for grid cells of 20 x 20 km² to enhance visibility. Presented results are for simulations under the RCP4.5 scenario and ‘high dispersal’ parameters set under four different levels of cultivation frequency (%).

Table 1: Linear regression of the number of cells simulated to be occupied in 2090 under the ‘high dispersal’ parameter set as a function of climate change scenario, cultivation frequency, and their interaction. Estimates of RCP-scenarios represent relative differences to the results obtained under constant climatic conditions. Lower AIC (Akaike Information Criterion) values indicate better models. Results under ‘low dispersal’ are similar (Supplementary Table S5).

| Predictors | Estimate | Std. error | <i>p</i> -value | AIC | R ² |
|---|----------|------------|-----------------|-------|----------------|
| Climate change scenario * Cultivation frequency | | | | 6684 | 0.62 |
| Baseline | 0.11 | 0.02 | <0.001 | | |
| RCP 2.6 | -0.31 | 0.03 | <0.001 | | |
| RCP 4.5 | 0.12 | 0.03 | <0.001 | | |
| RCP 8.5 | -0.27 | 0.03 | <0.001 | | |
| Cultivation frequency | 0.78 | 0.02 | <0.001 | | |
| RCP 2.6 : Cultivation frequency | -0.12 | 0.03 | <0.001 | | |
| RCP 4.5 : Cultivation frequency | 0.13 | 0.03 | <0.001 | | |
| RCP 8.5 : Cultivation frequency | -0.08 | 0.03 | <0.001 | | |
| excluding | | | | | |
| Climate change scenario | | | | 7047 | 0.58 |
| Cultivation frequency | | | | 10093 | 0.03 |
| Climate change scenario : Cultivation frequency | | | | 6762 | 0.62 |

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Authors' contributions: SD, MvK, OB, WT, FE conceived the ideas; GK, JW, AG, and DM downscaled the climatic data; ID measured seed traits and investigated species' habitat preferences; SR measured seed velocity; GK performed species distribution modelling and prepared input data for CATS; AG implemented CATS code in C++; AG and JW ran CATS simulations; GK analysed the modelling output; GK, FE and SD led the writing of the manuscript. All authors contributed substantially to the discussion of ideas and revised the text.

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APPENDICES

Abstract in German

Während der letzten zwei Jahrhunderte verursachte die Ausbreitung invasiver Arten (im Speziellen nicht heimische Arten, die anthropogen eingeführt werden, sich erfolgreich etablieren und weitreichend ausbreiten) negative Konsequenzen für die Biodiversität, Wirtschaft und das menschliche Wohlbefinden. Aufgrund dieser Entwicklung fokussierte sich die Wissenschaft zunehmend auf die Identifikation der Ursachen biologischer Invasionen, die Identifizierung zukünftiger Hochrisiko-Gebiete, und schließlich auf die Entwicklung von nationalen und internationalen Richtlinien zum Management von nicht heimischen Arten.

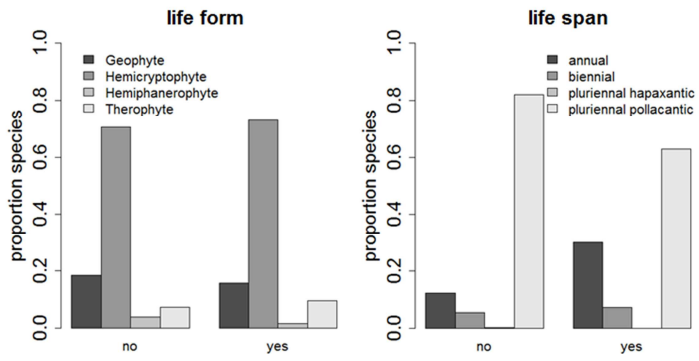
In meiner Dissertation begann ich mit Untersuchungen von Merkmalsprofilen hinsichtlich der Populationsbiologie und Verbreitung heimischer Pflanzenarten. Die Prognosefähigkeit, Arten, die in anderen Teilen der Welt als invasive Aliens gelten von solchen, die nirgendwo invasiv geworden sind, zu unterscheiden war nur gering. Die multidimensionale Variabilität im Merkmalsraum war jedoch größer in der Untergruppe der anderswo invasiven Aliens. Das zeigt, dass Invasionserfolg eine wichtige idiosynkratische Komponente hat und wahrscheinlich sehr kontextspezifisch ist.

Anhand von Artverbreitungsmodellen und gekoppelten Nischen- und Demografie-Modellen, habe ich des Weiteren das durch den Klimawandel veränderte Invasionsrisiko von Zierpflanzen in Europa erforscht, welche eine wichtige Quelle für potentiell zukünftige Invasoren darstellen. Die Zukunftsprojektionen zeigen eine starke Ausweitung an Gebieten mit einem erhöhten Risiko an erfolgreicher Etablierung von nicht heimischen Zierpflanzen. Trotzdem führt dieser Trend nicht zu einem Anstieg von Hybridisierungen zwischen heimischen Pflanzenarten und gattungsgleichen Zierpflanzen. Andererseits wird die durchschnittliche Fläche, die durch invasive Zierpflanzen am Ende des Jahrhunderts besiedelt sein wird, in den meisten Szenarien nicht größer, sondern kleiner als unter konstantem Klima. Einer der Hauptgründe ist die räumliche Verschiebung klimatisch geeigneter Flächen im Laufe des Jahrhunderts, welche vorübergehend etablierte Populationen wieder verschwinden lässt und dadurch eine vollständige Besiedlung geeigneter Gebiete verhindert. Meine Ergebnisse zeigen außerdem, dass Handelsbeschränkungen für potenziell invasive Zierpflanzen einen starken Effekt auf die Ausbreitung dieser Arten haben. Dieser Effekt ist nicht-linear, sodass positive Auswirkungen nur dann eintreten, wenn solche Beschränkungen rigoros umgesetzt werden.

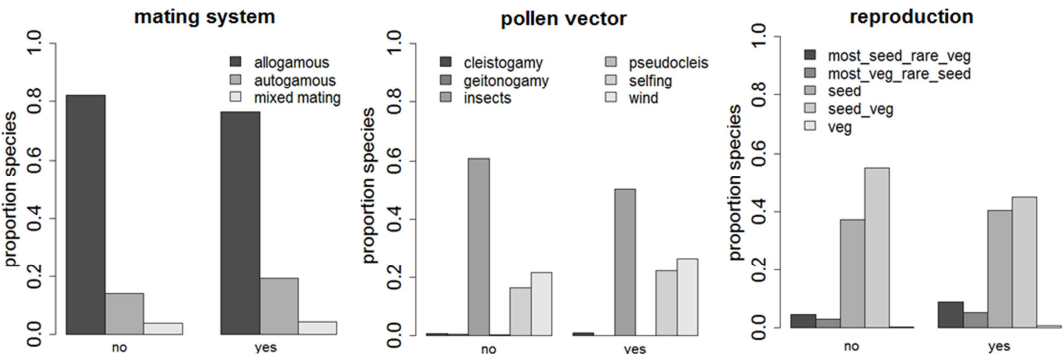
APPENDIX MANUSCRIPT 1

S1 Fig. Proportions of non-woody vascular plants of the Austrian flora that are invasive (“yes”) or not invasive (“no”) in other parts of the world in the different classes (= levels) of the analysed traits. Given are all 14 used traits in their corresponding trait group: (a) life history, (b) reproduction, (c) competitiveness, (d) habitat use, (e) propagule pressure.

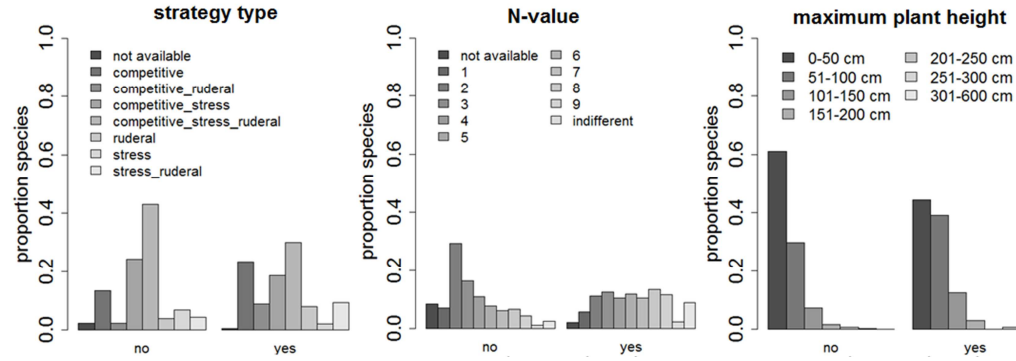
(a)



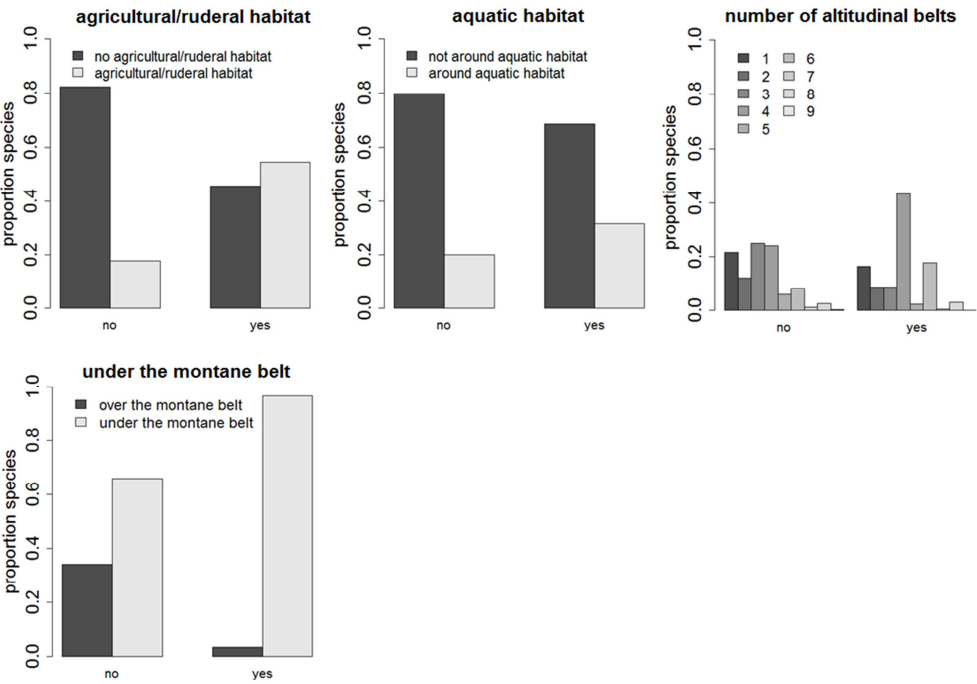
(b)



(c)

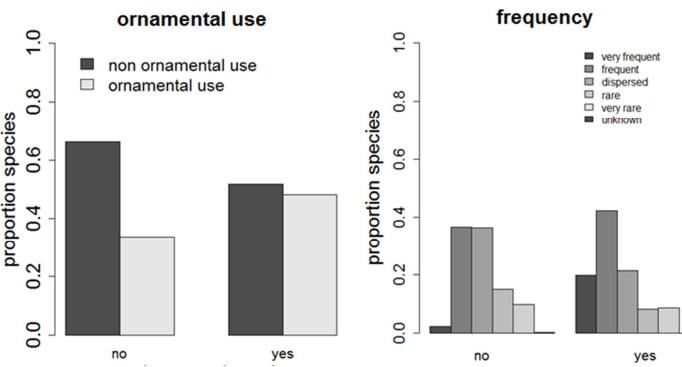


1 (d)



2

3 (e)



4

S1 Table. Categorization of ‘maximum plant height’.

| factor classes | metric values (cm) |
|-----------------------|---------------------------|
| 1 | 0-10 |
| 2 | 11-20 |
| 3 | 21-30 |
| 4 | 31-40 |
| 5 | 41-50 |
| 6 | 51-60 |
| 7 | 61-70 |
| 8 | 71-80 |
| 9 | 81-100 |
| 10 | 101-150 |
| 11 | 151-200 |
| 12 | 201-600 |

S2 Table. Candidate generalized linear mixed effects models (GLMM) evaluated based on the Akaike information criterion

(AIC). Numbers represent variable groups: (1) life history, (2) reproduction, (3) competitiveness, (4) habitat use, (5) propagule pressure. Variables within these groups were summarized by a multiple correspondence analysis and the first axis-values of these were used as predictors in the GLMMs. Bold markings refer to the most parsimonious model both without and with interactions.

Also given is a marginal R^2 [42, 43].

| <u>model</u> | R^2 | AIC | <u>model</u> | R^2 | AIC |
|--------------------------|--------------|----------------|--------------------------|--------------|----------------|
| no interactions | | | no interactions | | |
| 1 + 2 + 3 + 4 + 5 | 40.28 | 1134.20 | 2 + 3 | 7.30 | 1371.85 |
| 1 + 2 + 3 + 4 | 37.41 | 1178.69 | 2 | 0.51 | 1424.17 |
| 1 + 2 + 3 | 9.56 | 1356.25 | 2 + 4 + 5 | 37.65 | 1156.00 |
| 1 + 2 | 2.27 | 1411.19 | 2 + 4 | 35.47 | 1200.64 |
| 1 | 2.26 | 1409.23 | 2 + 5 | 10.60 | 1340.41 |
| 1 + 3 + 4 + 5 | 40.35 | 1133.00 | 3 + 4 + 5 | 39.28 | 1145.60 |
| 1 + 3 + 4 | 37.44 | 1176.95 | 3 + 4 | 37.20 | 1180.22 |
| 1 + 4 | 35.48 | 1198.88 | 3 + 5 | 13.11 | 1322.49 |
| 1 + 4 + 5 | 38.21 | 1148.46 | 3 | 6.04 | 1379.76 |
| 1 + 5 | 14.58 | 1312.03 | 4 + 5 | 37.61 | 1157.68 |
| 2 + 3 + 4 + 5 | 39.46 | 1141.73 | 4 | 35.52 | 1199.15 |
| 2 + 3 + 4 | 37.20 | 1180.08 | 5 | 9.17 | 1350.47 |
| with interactions | | | with interactions | | |
| 1*2+3+4+5 | 40.44 | 1135.99 | 2*3+4+5 | 40.12 | 1140.38 |
| 1*3+2+4+5 | 40.60 | 1135.39 | 2*3+4 | 37.73 | 1179.34 |
| 1*4+2+3+5 | 42.37 | 1131.71 | 2*3+1+5 | 19.36 | 1280.02 |
| 1*5+2+3+4 | 39.92 | 1134.96 | 2*3+5 | 15.48 | 1307.87 |
| 2*3+1+4+5 | 40.67 | 1133.33 | 2*3 | 7.55 | 1373.14 |
| 2*4+1+3+5 | 40.62 | 1135.53 | 2*4+1+3 | 37.44 | 1180.66 |
| 2*5+1+3+4 | 40.47 | 1135.89 | 2*4+1 | 35.50 | 1202.88 |
| 3*4+1+2+5 | 45.83 | 1129.91 | 2*4+3+5 | 39.65 | 1143.51 |
| 3*5+1+2+4 | 40.07 | 1135.15 | 2*4+3 | 37.21 | 1182.08 |
| 4*5+1+2+3 | 41.39 | 1135.78 | 2*4+1+5 | 38.49 | 1151.63 |
| 1*2+3*4+5 | 45.90 | 1131.71 | 2*4+5 | 37.88 | 1157.82 |
| 1*2+3*5+4 | 40.14 | 1136.92 | 2*4 | 35.46 | 1202.64 |
| 1*2+4*5+3 | 41.41 | 1137.56 | 2*5+1+3 | 19.15 | 1280.88 |
| 1*3+2*4+5 | 40.83 | 1136.78 | 2*5+1 | 14.63 | 1315.06 |
| 1*3+2*5+4 | 40.65 | 1137.14 | 2*5+3+4 | 39.68 | 1143.32 |
| 1*3+4*5+2 | 41.51 | 1137.00 | 2*5+3 | 15.26 | 1308.75 |
| 1*4+2*3+5 | 42.80 | 1131.22 | 2*5+1+4 | 38.28 | 1152.02 |
| 1*4+2*5+3 | 42.22 | 1133.60 | 2*5+4 | 37.80 | 1157.76 |
| 1*4+3*5+2 | 42.10 | 1132.79 | 2*5 | 10.63 | 1342.39 |
| 1*5+2*3+4 | 40.33 | 1133.99 | 3*4+1+2 | 44.44 | 1172.82 |
| 1*5+2*4+3 | 40.20 | 1136.00 | 3*4+1 | 44.49 | 1171.11 |
| 1*5+3*4+2 | 45.45 | 1130.75 | 3*4+2+5 | 45.84 | 1136.21 |
| 2*3+4*5+1 | 41.98 | 1134.87 | 3*4+2 | 44.62 | 1173.52 |
| 2*4+3*5+1 | 40.31 | 1136.60 | 3*4+1+5 | 45.92 | 1128.75 |
| 2*5+3*4+1 | 46.12 | 1131.48 | 3*4+5 | 46.08 | 1139.76 |
| 1*2+3+4 | 37.58 | 1179.83 | 3*4 | 44.80 | 1173.51 |
| 1*2+3 | 9.64 | 1357.08 | 3*5+1+2 | 19.12 | 1280.88 |
| 1*2+4+5 | 38.20 | 1152.06 | 3*5+1 | 18.89 | 1281.51 |
| 1*2+4 | 35.62 | 1202.17 | 3*5+2+4 | 39.29 | 1141.90 |
| 1*2+3+5 | 19.16 | 1280.51 | 3*5+2 | 14.98 | 1308.63 |
| 1*2+5 | 14.66 | 1314.72 | 3*5+1+4 | 40.12 | 1133.93 |
| 1*2 | 2.38 | 1411.84 | 3*5+4 | 39.06 | 1145.36 |
| 1*3+2+4 | 37.59 | 1180.28 | 3*5 | 12.77 | 1324.12 |
| 1*3+2 | 9.75 | 1357.78 | 4*5+1+2 | 39.59 | 1151.63 |
| 1*3+4+5 | 40.70 | 1134.16 | 4*5+1 | 39.59 | 1149.94 |
| 1*3+4 | 37.63 | 1178.53 | 4*5+2+3 | 41.04 | 1143.01 |
| 1*3+2+5 | 19.39 | 1280.22 | 4*5+2 | 39.47 | 1157.15 |
| 1*3+5 | 19.19 | 1280.86 | 4*5+1+3 | 41.39 | 1134.64 |
| 1*3 | 9.58 | 1357.13 | 4*5+3 | 40.81 | 1146.96 |
| 1*4+2+3 | 39.40 | 1177.11 | 4*5 | 39.38 | 1158.90 |
| 1*4+2 | 37.75 | 1199.03 | 1*2+3*4 | 44.65 | 1173.94 |
| 1*4+3+5 | 42.40 | 1130.79 | 1*2+3*5 | 19.14 | 1282.51 |

| | | | | | |
|---------|-------|---------|---------|-------|---------|
| 1*4+3 | 39.40 | 1175.50 | 1*2+4*5 | 39.64 | 1153.50 |
| 1*4+2+5 | 40.59 | 1147.50 | 1*3+2*4 | 37.62 | 1182.26 |
| 1*4+5 | 40.61 | 1145.96 | 1*3+2*5 | 19.38 | 1282.22 |
| 1*4 | 37.76 | 1197.05 | 1*3+4*5 | 41.77 | 1135.81 |
| 1*5+2+3 | 18.85 | 1278.84 | 1*4+2*3 | 39.71 | 1177.03 |
| 1*5+2 | 14.32 | 1312.16 | 1*4+2*5 | 40.62 | 1149.48 |
| 1*5+3+4 | 39.94 | 1133.51 | 1*4+3*5 | 42.15 | 1131.82 |
| 1*5+3 | 18.64 | 1278.98 | 1*5+2*3 | 19.04 | 1279.77 |
| 1*5+2+4 | 37.78 | 1150.19 | 1*5+2*4 | 38.14 | 1151.30 |
| 1*5+4 | 37.80 | 1148.32 | 1*5+3*4 | 45.52 | 1129.32 |
| 1*5 | 14.25 | 1310.81 | 2*3+4*5 | 41.68 | 1141.61 |
| 2*3+1+4 | 37.86 | 1178.19 | 2*4+3*5 | 39.39 | 1143.75 |
| 2*3+1 | 9.77 | 1357.55 | 2*5+3*4 | 46.17 | 1137.66 |

S3 Table. List of study species analysed and their status of being invasive somewhere outside their native distribution. The species are first ordered according to their status of being invasive outside of their native range, after a classification of species as environmental weeds in the latest edition of Randall's Global Compendium of Weeds [38]. Further species are listed alphabetically.

| Species name env.weed | | | | | |
|---------------------------|---|------------------------|---|-------------------------|---|
| Achillea millefolium | 1 | Cardaminopsis arenosa | 0 | Lychnis viscaria | 0 |
| Achillea nobilis | 1 | Cardaminopsis halleri | 0 | Lycopus europaeus | 0 |
| Achillea ptarmica | 1 | Cardaminopsis petraea | 0 | Lycopus exaltatus | 0 |
| Achnatherum calamagrostis | 1 | Carduus defloratus | 0 | Maianthemum bifolium | 0 |
| Acinos arvensis | 1 | Carduus personata | 0 | Malaxis monophyllos | 0 |
| Aconitum napellus | 1 | Carex acuta | 0 | Melampyrum arvense | 0 |
| Adoxa moschatellina | 1 | Carex acutiformis | 0 | Melampyrum cristatum | 0 |
| Aegopodium podagraria | 1 | Carex alba | 0 | Melampyrum nemorosum | 0 |
| Agrostis canina | 1 | Carex appropinquata | 0 | Melampyrum pratense | 0 |
| Agrostis capillaris | 1 | Carex atrata | 0 | Melampyrum sylvaticum | 0 |
| Agrostis gigantea | 1 | Carex baldensis | 0 | Melica ciliata | 0 |
| Agrostis stolonifera | 1 | Carex bigelowii | 0 | Melica picta | 0 |
| Aira caryophylla | 1 | Carex bohémica | 0 | Melica transsilvanica | 0 |
| Ajuga genevensis | 1 | Carex brachystachys | 0 | Melica uniflora | 0 |
| Ajuga reptans | 1 | Carex brunnescens | 0 | Melilotus altissimus | 0 |
| Allium carinatum | 1 | Carex buekii | 0 | Melilotus dentatus | 0 |
| Allium oleraceum | 1 | Carex buxbaumii | 0 | Melittis melissophyllum | 0 |
| Allium scorodoprasum | 1 | Carex canescens | 0 | Mentha dumetorum | 0 |
| Allium vineale | 1 | Carex capillaris | 0 | Mentha longifolia | 0 |
| Alopecurus aequalis | 1 | Carex caryophylla | 0 | Meum athamanticum | 0 |
| Alopecurus geniculatus | 1 | Carex cespitosa | 0 | Milium effusum | 0 |
| Alopecurus pratensis | 1 | Carex chordorrhiza | 0 | Minuartia hybrida | 0 |
| Alyssum alyssoides | 1 | Carex curvata | 0 | Minuartia rubra | 0 |
| Angelica sylvestris | 1 | Carex davalliana | 0 | Minuartia viscosa | 0 |
| Anthoxanthum odoratum | 1 | Carex demissa | 0 | Moehringia ciliata | 0 |
| Anthriscus sylvestris | 1 | Carex diandra | 0 | Moehringia muscosa | 0 |
| Apium graveolens | 1 | Carex digitata | 0 | Moehringia trinervia | 0 |
| Apium nodiflorum | 1 | Carex dioica | 0 | Molinia arundinacea | 0 |
| Aquilegia vulgaris | 1 | Carex distans | 0 | Monotropa hypophegea | 0 |
| Arctium minus | 1 | Carex echinata | 0 | Monotropa hypopitys | 0 |
| Arenaria leptoclados | 1 | Carex elata | 0 | Montia fontana | 0 |
| Arenaria serpyllifolia | 1 | Carex elongata | 0 | Muscari tenuiflorum | 0 |
| Arrhenatherum elatius | 1 | Carex ericetorum | 0 | Myosotis alpestris | 0 |
| Artemisia vulgaris | 1 | Carex ferruginea | 0 | Myosotis decumbens | 0 |
| Aruncus dioicus | 1 | Carex firma | 0 | Myosotis nemorosa | 0 |
| Atriplex prostrata | 1 | Carex frigida | 0 | Myosotis rehsteineri | 0 |
| Barbarea vulgaris | 1 | Carex fuliginosa | 0 | Myosotis sparsiflora | 0 |
| Bidens cernua | 1 | Carex halleriana | 0 | Narcissus radiiflorus | 0 |
| Bidens radiata | 1 | Carex hartmanii | 0 | Neottia nidus-avis | 0 |
| Bidens tripartita | 1 | Carex heleonastes | 0 | Nigritella nigra | 0 |
| Bothriochloa ischaemum | 1 | Carex hirta | 0 | Nigritella rhellicani | 0 |
| Brachypodium pinnatum | 1 | Carex hordeistichos | 0 | Nigritella rubra | 0 |
| Brachypodium sylvaticum | 1 | Carex hostiana | 0 | Nigritella widderi | 0 |
| Bromus erectus | 1 | Carex humilis | 0 | Nonea pulla | 0 |
| Bromus hordeaceus | 1 | Carex limosa | 0 | Odontites luteus | 0 |
| Bromus inermis | 1 | Carex melanostachya | 0 | Odontites vernus | 0 |
| Bromus racemosus | 1 | Carex michelii | 0 | Odontites vulgaris | 0 |
| Bryonia dioica | 1 | Carex microglochin | 0 | Omphalodes scorpioides | 0 |
| Calamagrostis epigejos | 1 | Carex montana | 0 | Onobrychis arenaria | 0 |
| Calystegia sepium | 1 | Carex mucronata | 0 | Onobrychis montana | 0 |
| Campanula rapunculoides | 1 | Carex muricata | 0 | Onosma arenaria | 0 |
| Capsella bursa-pastoris | 1 | Carex nigra | 0 | Ophrys apifera | 0 |
| Cardamine flexuosa | 1 | Carex oenensis | 0 | Ophrys araneola | 0 |
| Cardamine hirsuta | 1 | Carex ornithopoda | 0 | Ophrys holoserica | 0 |
| Cardamine impatiens | 1 | Carex ornithopodioides | 0 | Ophrys insectifera | 0 |

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|-------------------------|---|------------------------------|---|--------------------------------|---|
| Cardamine pratensis | 1 | Carex otrubae | 0 | Ophrys sphegodes | 0 |
| Carduus crispus | 1 | Carex pairae | 0 | Orchis mascula | 0 |
| Carex brizoides | 1 | Carex pallescens | 0 | Orchis militaris | 0 |
| Carex disticha | 1 | Carex parviflora | 0 | Orchis pallens | 0 |
| Carex divulsa | 1 | Carex pauciflora | 0 | Orchis palustris | 0 |
| Carex flacca | 1 | Carex paupercula | 0 | Orchis purpurea | 0 |
| Carex flava | 1 | Carex pendula | 0 | Orchis simia | 0 |
| Carex lepidocarpa | 1 | Carex pilosa | 0 | Orchis spitzelii | 0 |
| Carex ovalis | 1 | Carex praecox | 0 | Orchis tridentata | 0 |
| Carex panicea | 1 | Carex pulcaris | 0 | Orchis ustulata | 0 |
| Carex paniculata | 1 | Carex randalpina | 0 | Oreochloa disticha | 0 |
| Carex pilulifera | 1 | Carex remota | 0 | Ornithogalum pyrenaicum | 0 |
| Carex punctata | 1 | Carex rostrata | 0 | Oxalis acetosella | 0 |
| Carlina vulgaris | 1 | Carex rupestris | 0 | Oxyria digyna | 0 |
| Carpesium cernuum | 1 | Carex secalina | 0 | Oxytropis jacquinii | 0 |
| Carum carvi | 1 | Carex sempervirens | 0 | Oxytropis pilosa | 0 |
| Catabrosa aquatica | 1 | Carex spicata | 0 | Papaver alpinum | 0 |
| Centaurea jacea | 1 | Carex strigosa | 0 | Paris quadrifolia | 0 |
| Centaurea montana | 1 | Carex supina | 0 | Parnassia palustris | 0 |
| Centaurea stoebe | 1 | Carex sylvatica | 0 | Pedicularis elongata | 0 |
| Centaurium erythraea | 1 | Carex tomentosa | 0 | Pedicularis foliosa | 0 |
| Centaurium pulchellum | 1 | Carex umbrosa | 0 | Pedicularis oederi | 0 |
| Cerastium diffusum | 1 | Carex vaginata | 0 | Pedicularis palustris | 0 |
| Cerastium dubium | 1 | Carex viridula | 0 | Pedicularis recutita | 0 |
| Cerastium semidecandrum | 1 | Carex vulpina | 0 | Pedicularis rostratocapitata | 0 |
| Chelidonium majus | 1 | Carlina acaulis | 0 | Pedicularis rostratospicata | 0 |
| Chenopodium polyspermum | 1 | Carlina biebersteinii | 0 | Pedicularis sceptrum-carolinum | 0 |
| Cirsium arvense | 1 | Centaurea phrygia | 0 | Pedicularis sylvatica | 0 |
| Cirsium oleraceum | 1 | Centaurea pseudophrygia | 0 | Pedicularis verticillata | 0 |
| Cirsium palustre | 1 | Centaurea scabiosa | 0 | Peplis portula | 0 |
| Cirsium vulgare | 1 | Centaurea stenolepis | 0 | Persicaria amphibia | 0 |
| Clinopodium vulgare | 1 | Centaurea triumfettii | 0 | Persicaria brittingeri | 0 |
| Convallaria majalis | 1 | Centaurium littorale | 0 | Persicaria dubia | 0 |
| Convolvulus arvensis | 1 | Centunculus minimus | 0 | Persicaria hydropiper | 0 |
| Corynephorus canescens | 1 | Cephalanthera damasonium | 0 | Persicaria minor | 0 |
| Crepis tectorum | 1 | Cephalanthera longifolia | 0 | Petasites albus | 0 |
| Cuscuta epithymum | 1 | Cephalanthera rubra | 0 | Petasites hybridus | 0 |
| Cynoglossum officinale | 1 | Cerastium glutinosum | 0 | Petasites paradoxus | 0 |
| Cynosurus cristatus | 1 | Cerastium pumilum | 0 | Peucedanum alsaticum | 0 |
| Cyperus longus | 1 | Ceratocarpus claviculata | 0 | Peucedanum carvifolia | 0 |
| Dactylis glomerata | 1 | Cerinthe glabra | 0 | Peucedanum cervaria | 0 |
| Danthonia decumbens | 1 | Chaerophyllum aromaticum | 0 | Peucedanum officinale | 0 |
| Daucus carota | 1 | Chaerophyllum aureum | 0 | Peucedanum oreoselinum | 0 |
| Deschampsia flexuosa | 1 | Chaerophyllum bulbosum | 0 | Peucedanum ostruthium | 0 |
| Dianthus armeria | 1 | Chaerophyllum hirsutum | 0 | Peucedanum palustre | 0 |
| Elatine triandra | 1 | Chaerophyllum temulum | 0 | Phleum alpinum | 0 |
| Eleocharis palustris | 1 | Chamorchis alpina | 0 | Phleum bertolonii | 0 |
| Elytrigia intermedia | 1 | Chenopodium bonus-henricus | 0 | Phleum hirsutum | 0 |
| Elytrigia repens | 1 | Chenopodium botryodes | 0 | Phleum paniculatum | 0 |
| Epilobium angustifolium | 1 | Chenopodium rubrum | 0 | Phleum phleoides | 0 |
| Epilobium hirsutum | 1 | Chimaphila umbellata | 0 | Phleum rhaeticum | 0 |
| Epipactis helleborine | 1 | Chrysosplenium alternifolium | 0 | Phyteuma betonicifolium | 0 |
| Erica tetralix | 1 | Cicerbita alpina | 0 | Phyteuma hemisphaericum | 0 |
| Erodium cicutarium | 1 | Cicuta virosa | 0 | Phyteuma nigrum | 0 |
| Erophila verna | 1 | Circaea alpina | 0 | Phyteuma orbiculare | 0 |
| Erysimum cheiranthoides | 1 | Circaea intermedia | 0 | Phyteuma ovatum | 0 |
| Erysimum repandum | 1 | Circaea lutetiana | 0 | Phyteuma spicatum | 0 |
| Eupatorium cannabinum | 1 | Cirsium acaule | 0 | Pimpinella major | 0 |
| Euphorbia cyparissias | 1 | Cirsium canum | 0 | Pimpinella nigra | 0 |
| Euphorbia epithymoides | 1 | Cirsium eriophorum | 0 | Pimpinella saxifraga | 0 |
| Euphorbia esula | 1 | Cirsium helenioides | 0 | Pinguicula alpina | 0 |
| Euphorbia platyphyllos | 1 | Cirsium rivulare | 0 | Pinguicula vulgaris | 0 |

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|--------------------------|---|----------------------------|---|-------------------------------|---|
| Filipendula ulmaria | 1 | Cirsium spinosissimum | 0 | Plantago alpina | 0 |
| Fragaria vesca | 1 | Cirsium tuberosum | 0 | Plantago atrata | 0 |
| Galanthus nivalis | 1 | Clematis recta | 0 | Plantago intermedia | 0 |
| Galeopsis tetrahit | 1 | Cochlearia pyrenaica | 0 | Plantago maritima | 0 |
| Galium aparine | 1 | Coeloglossum viride | 0 | Plantago media | 0 |
| Galium palustre | 1 | Colchicum autumnale | 0 | Plantago strictissima | 0 |
| Galium verum | 1 | Coleanthus subtilis | 0 | Plantago winteri | 0 |
| Genista tinctoria | 1 | Corallorrhiza trifida | 0 | Platanthera bifolia | 0 |
| Geranium lucidum | 1 | Cortusa matthioli | 0 | Platanthera chlorantha | 0 |
| Geranium robertianum | 1 | Corydalis cava | 0 | Pleurospermum austriacum | 0 |
| Geranium sanguineum | 1 | Corydalis intermedia | 0 | Poa alpina | 0 |
| Glechoma hederacea | 1 | Corydalis pumila | 0 | Poa angustifolia | 0 |
| Glyceria declinata | 1 | Corydalis solida | 0 | Poa badensis | 0 |
| Holcus lanatus | 1 | Crassula aquatica | 0 | Poa cenisia | 0 |
| Holcus mollis | 1 | Crepis alpestris | 0 | Poa chaixii | 0 |
| Holosteum umbellatum | 1 | Crepis aurea | 0 | Poa glauca | 0 |
| Humulus lupulus | 1 | Crepis bocconi | 0 | Poa humilis | 0 |
| Hypericum humifusum | 1 | Crepis conyzifolia | 0 | Poa hybrida | 0 |
| Hypericum perforatum | 1 | Crepis jacquini | 0 | Poa minor | 0 |
| Hypericum tetrapterum | 1 | Crepis mollis | 0 | Poa nemoralis | 0 |
| Hypochaeris glabra | 1 | Crepis paludosa | 0 | Poa remota | 0 |
| Hypochaeris radicata | 1 | Crepis praemorsa | 0 | Poa supina | 0 |
| Illecebrum verticillatum | 1 | Crepis pyrenaica | 0 | Podospermum laciniatum | 0 |
| Impatiens noli-tangere | 1 | Crepis terglouensis | 0 | Polemonium caeruleum | 0 |
| Iris pseudacorus | 1 | Crocus albiflorus | 0 | Polycnemum verrucosum | 0 |
| Iris spuria | 1 | Cruciata glabra | 0 | Polygala alpestris | 0 |
| Juncus acutiflorus | 1 | Cucubalus baccifer | 0 | Polygala chamaebuxus | 0 |
| Juncus articulatus | 1 | Cuscuta europaea | 0 | Polygala comosa | 0 |
| Juncus bufonius | 1 | Cuscuta lupuliformis | 0 | Polygonatum multiflorum | 0 |
| Juncus capitatus | 1 | Cyclamen purpurascens | 0 | Polygonatum odoratum | 0 |
| Juncus compressus | 1 | Cynoglossum germanicum | 0 | Polygonatum verticillatum | 0 |
| Juncus effusus | 1 | Cyperus flavescens | 0 | Potentilla alba | 0 |
| Juncus gerardii | 1 | Cyperus fuscus | 0 | Potentilla anglica | 0 |
| Juncus inflexus | 1 | Cypripedium calceolus | 0 | Potentilla anserina | 0 |
| Juncus maritimus | 1 | Dactylis polygama | 0 | Potentilla aurea | 0 |
| Juncus squarrosus | 1 | Dactylorhiza cruenta | 0 | Potentilla brauneana | 0 |
| Knautia arvensis | 1 | Dactylorhiza fuchsii | 0 | Potentilla caulescens | 0 |
| Lactuca saligna | 1 | Dactylorhiza incarnata | 0 | Potentilla clusiana | 0 |
| Lactuca serriola | 1 | Dactylorhiza lapponica | 0 | Potentilla collina | 0 |
| Lactuca virosa | 1 | Dactylorhiza maculata | 0 | Potentilla crantzii | 0 |
| Lamium maculatum | 1 | Dactylorhiza majalis | 0 | Potentilla incana | 0 |
| Lapsana communis | 1 | Dactylorhiza sambucina | 0 | Potentilla micrantha | 0 |
| Lathyrus sylvestris | 1 | Dactylorhiza traunsteineri | 0 | Potentilla mixta | 0 |
| Leontodon autumnalis | 1 | Danthonia alpina | 0 | Potentilla neglecta | 0 |
| Leontodon hispidus | 1 | Daphne cneorum | 0 | Potentilla pusilla | 0 |
| Leontodon saxatilis | 1 | Daphne striata | 0 | Potentilla rupestris | 0 |
| Lepidium latifolium | 1 | Dentaria bulbifera | 0 | Potentilla sterilis | 0 |
| Leucanthemum vulgare | 1 | Dentaria enneaphyllos | 0 | Potentilla thuringiaca | 0 |
| Linaria vulgaris | 1 | Dentaria pentaphyllos | 0 | Potentilla thyrsiflora | 0 |
| Linum catharticum | 1 | Deschampsia cespitosa | 0 | Prenanthes purpurea | 0 |
| Lolium perenne | 1 | Deschampsia littoralis | 0 | Primula auricula | 0 |
| Lotus corniculatus | 1 | Dianthus carthusianorum | 0 | Primula clusiana | 0 |
| Lotus pedunculatus | 1 | Dianthus deltoides | 0 | Primula elatior | 0 |
| Ludwigia palustris | 1 | Dianthus seguieri | 0 | Primula farinosa | 0 |
| Lunaria rediviva | 1 | Dianthus superbus | 0 | Primula hirsuta | 0 |
| Luzula luzuloides | 1 | Dianthus sylvestris | 0 | Primula minima | 0 |
| Luzula sylvatica | 1 | Dichostylis micheliana | 0 | Primula veris | 0 |
| Lychnis flos-cuculi | 1 | Dictamnus albus | 0 | Primula vulgaris | 0 |
| Lysimachia nummularia | 1 | Digitalis grandiflora | 0 | Pseudolysimachion longifolium | 0 |
| Lysimachia thyrsiflora | 1 | Digitalis lutea | 0 | Pseudorchis albida | 0 |
| Lysimachia vulgaris | 1 | Doronicum austriacum | 0 | Psilathera ovata | 0 |
| Lythrum hyssopifolia | 1 | Doronicum columnae | 0 | Puccinellia distans | 0 |

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| Lythrum salicaria | 1 | Doronicum glaciale | 0 | Puccinellia limosa | 0 |
| Medicago falcata | 1 | Doronicum grandiflorum | 0 | Pulicaria dysenterica | 0 |
| Medicago lupulina | 1 | Doronicum pardalianches | 0 | Pulicaria vulgaris | 0 |
| Medicago minima | 1 | Dorycnium germanicum | 0 | Pulmonaria angustifolia | 0 |
| Melica nutans | 1 | Dorycnium herbaceum | 0 | Pulmonaria mollis | 0 |
| Mentha aquatica | 1 | Drosera intermedia | 0 | Pulmonaria obscura | 0 |
| Mentha arvensis | 1 | Drosera longifolia | 0 | Pulmonaria officinalis | 0 |
| Mentha pulegium | 1 | Drosera obovata | 0 | Pulsatilla alpina | 0 |
| Mentha verticillata | 1 | Drosera rotundifolia | 0 | Pulsatilla micrantha | 0 |
| Mercurialis perennis | 1 | Dryas octopetala | 0 | Pulsatilla pratensis | 0 |
| Molinia caerulea | 1 | Elatine hexandra | 0 | Pulsatilla vernalis | 0 |
| Muscari botryoides | 1 | Elatine hydropiper | 0 | Pulsatilla vulgaris | 0 |
| Mycelis muralis | 1 | Eleocharis acicularis | 0 | Pyrola chlorantha | 0 |
| Myosotis discolor | 1 | Eleocharis austriaca | 0 | Pyrola media | 0 |
| Myosotis laxa | 1 | Eleocharis mamillata | 0 | Pyrola minor | 0 |
| Myosotis scorpioides | 1 | Eleocharis ovata | 0 | Pyrola rotundifolia | 0 |
| Myosotis sylvatica | 1 | Eleocharis quinqueflora | 0 | Radiola linoides | 0 |
| Myosoton aquaticum | 1 | Eleocharis uniglumis | 0 | Ranunculus aconitifolius | 0 |
| Narcissus pseudonarcissus | 1 | Elyna myosuroides | 0 | Ranunculus alpestris | 0 |
| Nardus stricta | 1 | Elytrigia atherica | 0 | Ranunculus argoviensis | 0 |
| Ononis repens | 1 | Empetrum hermaphroditum | 0 | Ranunculus breyninus | 0 |
| Ononis spinosa | 1 | Empetrum nigrum | 0 | Ranunculus carinthiacus | 0 |
| Origanum vulgare | 1 | Epilobium alpestre | 0 | Ranunculus cassubicifolius | 0 |
| Pastinaca sativa | 1 | Epilobium alsinifolium | 0 | Ranunculus dactylophyllus | 0 |
| Persicaria lapathifolia | 1 | Epilobium anagallidifolium | 0 | Ranunculus glacialis | 0 |
| Persicaria maculosa | 1 | Epilobium collinum | 0 | Ranunculus hybridus | 0 |
| Petrorhagia prolifera | 1 | Epilobium lamyi | 0 | Ranunculus illyricus | 0 |
| Phalaris arundinacea | 1 | Epilobium lanceolatum | 0 | Ranunculus lanuginosus | 0 |
| Phleum pratense | 1 | Epilobium montanum | 0 | Ranunculus megacarpus | 0 |
| Physalis alkekengi | 1 | Epilobium nutans | 0 | Ranunculus montanus | 0 |
| Picris hieracioides | 1 | Epilobium obscurum | 0 | Ranunculus nemorosus | 0 |
| Plantago major | 1 | Epilobium palustre | 0 | Ranunculus parnassiifolius | 0 |
| Poa annua | 1 | Epilobium parviflorum | 0 | Ranunculus phragmiteti | 0 |
| Poa bulbosa | 1 | Epilobium roseum | 0 | Ranunculus platanifolius | 0 |
| Poa compressa | 1 | Epilobium tetragonum | 0 | Ranunculus polyanthemoides | 0 |
| Poa palustris | 1 | Epipactis albensis | 0 | Ranunculus polyanthemophyllus | 0 |
| Poa pratensis | 1 | Epipactis atrorubens | 0 | Ranunculus polyanthemus | 0 |
| Poa trivialis | 1 | Epipactis greuteri | 0 | Ranunculus reptans | 0 |
| Polygonum arenastrum | 1 | Epipactis leptochila | 0 | Ranunculus serpens | 0 |
| Polygonum aviculare | 1 | Epipactis microphylla | 0 | Ranunculus villarsii | 0 |
| Potentilla argentea | 1 | Epipactis muelleri | 0 | Rapistrum perenne | 0 |
| Potentilla erecta | 1 | Epipactis palustris | 0 | Rhamnus pumila | 0 |
| Potentilla reptans | 1 | Epipogium aphyllum | 0 | Rhinanthus alectorolophus | 0 |
| Potentilla supina | 1 | Erica carnea | 0 | Rhinanthus angustifolius | 0 |
| Pseudognaphalium luteoalbum | 1 | Erigeron acris | 0 | Rhinanthus glacialis | 0 |
| Ranunculus acris | 1 | Erigeron alpinus | 0 | Rhinanthus minor | 0 |
| Ranunculus bulbosus | 1 | Erigeron atticus | 0 | Rhinanthus pulcher | 0 |
| Ranunculus ficaria | 1 | Erigeron glabratus | 0 | Rhodiola rosea | 0 |
| Ranunculus flammula | 1 | Erigeron neglectus | 0 | Rhodothamnus chamaecistus | 0 |
| Ranunculus lingua | 1 | Erigeron uniflorus | 0 | Rhynchospora alba | 0 |
| Ranunculus repens | 1 | Eriophorum angustifolium | 0 | Rhynchospora fusca | 0 |
| Ranunculus sardous | 1 | Eriophorum gracile | 0 | Roegneria canina | 0 |
| Ranunculus sceleratus | 1 | Eriophorum latifolium | 0 | Rorippa pyrenaica | 0 |
| Rorippa amphibia | 1 | Eriophorum scheuchzeri | 0 | Rumex alpinus | 0 |
| Rorippa austriaca | 1 | Eriophorum vaginatum | 0 | Rumex aquaticus | 0 |
| Rorippa palustris | 1 | Erucastrum nasturtiifolium | 0 | Rumex arifolius | 0 |
| Rorippa sylvestris | 1 | Eryngium campestre | 0 | Rumex hydrolapathum | 0 |
| Rumex acetosa | 1 | Eryngium planum | 0 | Rumex maritimus | 0 |
| Rumex acetosella | 1 | Erysimum marschallianum | 0 | Rumex palustris | 0 |
| Rumex conglomeratus | 1 | Erysimum odoratum | 0 | Rumex thyrsiflorus | 0 |
| Rumex crispus | 1 | Erysimum virgatum | 0 | Sagina nodosa | 0 |
| Rumex obtusifolius | 1 | Euphorbia dulcis | 0 | Sagina normaniana | 0 |

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|-----------------------------|---|--------------------------|---|---------------------------|---|
| Rumex sanguineus | 1 | Euphorbia lucida | 0 | Sagina saginoides | 0 |
| Salsola kali | 1 | Euphorbia palustris | 0 | Salicornia europaea | 0 |
| Salvia nemorosa | 1 | Euphorbia salicifolia | 0 | Salix alpina | 0 |
| Salvia pratensis | 1 | Euphorbia seguieriana | 0 | Salix breviserrata | 0 |
| Sambucus ebulus | 1 | Euphorbia stricta | 0 | Salix herbacea | 0 |
| Samolus valerandi | 1 | Euphorbia verrucosa | 0 | Salix myrtilloides | 0 |
| Sanguisorba minor | 1 | Euphorbia villosa | 0 | Salix reticulata | 0 |
| Saponaria ocymoides | 1 | Euphrasia hirtella | 0 | Salix retusa | 0 |
| Saponaria officinalis | 1 | Euphrasia micrantha | 0 | Salix serpyllifolia | 0 |
| Sclerochloa dura | 1 | Euphrasia minima | 0 | Salvia glutinosa | 0 |
| Securigera varia | 1 | Euphrasia nemorosa | 0 | Sanguisorba officinalis | 0 |
| Sedum telephium | 1 | Euphrasia officinalis | 0 | Sanicula europaea | 0 |
| Senecio jacobaea | 1 | Euphrasia salisburgensis | 0 | Saussurea alpina | 0 |
| Senecio sylvaticus | 1 | Euphrasia stricta | 0 | Saussurea discolor | 0 |
| Senecio vulgaris | 1 | Euphrasia tricuspidata | 0 | Saussurea pygmaea | 0 |
| Silene conica | 1 | Falcaria vulgaris | 0 | Saxifraga granulata | 0 |
| Silene latifolia | 1 | Fallopia dumetorum | 0 | Saxifraga rotundifolia | 0 |
| Solanum dulcamara | 1 | Filago arvensis | 0 | Saxifraga tridactylites | 0 |
| Sonchus arvensis | 1 | Filago lutescens | 0 | Scabiosa canescens | 0 |
| Sonchus asper | 1 | Filago minima | 0 | Scabiosa columbaria | 0 |
| Sonchus oleraceus | 1 | Filago vulgaris | 0 | Scabiosa lucida | 0 |
| Sonchus palustris | 1 | Filipendula vulgaris | 0 | Scabiosa ochroleuca | 0 |
| Spergula morisonii | 1 | Fragaria moschata | 0 | Scheuchzeria palustris | 0 |
| Spergula pentandra | 1 | Fragaria viridis | 0 | Schoenoplectus mucronatus | 0 |
| Spergularia media | 1 | Fumana procumbens | 0 | Schoenoplectus pungens | 0 |
| Stachys palustris | 1 | Gagea bohemica | 0 | Schoenoplectus supinus | 0 |
| Stellaria alsine | 1 | Gagea fragifera | 0 | Schoenoplectus triqueter | 0 |
| Stellaria graminea | 1 | Gagea lutea | 0 | Schoenus ferrugineus | 0 |
| Suaeda maritima | 1 | Gagea minima | 0 | Schoenus nigricans | 0 |
| Symphytum officinale | 1 | Gagea pomeranica | 0 | Scilla bifolia | 0 |
| Tamus communis | 1 | Gagea pratensis | 0 | Scirpoides holoschoenus | 0 |
| Teesdalia nudicaulis | 1 | Gagea spathacea | 0 | Scirpus radicans | 0 |
| Thlaspi perfoliatum | 1 | Galeopsis bifida | 0 | Scirpus sylvaticus | 0 |
| Thymelaea passerina | 1 | Galeopsis ladanum | 0 | Scleranthus perennis | 0 |
| Thymus pulegioides | 1 | Galeopsis pubescens | 0 | Scleranthus polycarpus | 0 |
| Thymus serpyllum | 1 | Galeopsis speciosa | 0 | Scleranthus verticillatus | 0 |
| Torilis japonica | 1 | Galium album | 0 | Scorzonera austriaca | 0 |
| Tragopogon dubius | 1 | Galium aristatum | 0 | Scorzonera hispanica | 0 |
| Tragopogon pratensis | 1 | Galium boreale | 0 | Scorzonera humilis | 0 |
| Trifolium arvense | 1 | Galium glaucum | 0 | Scorzonera parviflora | 0 |
| Trifolium aureum | 1 | Galium lucidum | 0 | Scorzonera purpurea | 0 |
| Trifolium campestre | 1 | Galium megalospermum | 0 | Scrophularia canina | 0 |
| Trifolium dubium | 1 | Galium pomeranicum | 0 | Scrophularia nodosa | 0 |
| Trifolium fragiferum | 1 | Galium pumilum | 0 | Scrophularia umbrosa | 0 |
| Trifolium hybridum | 1 | Galium schultesii | 0 | Scutellaria galericulata | 0 |
| Trifolium pratense | 1 | Galium sternerii | 0 | Scutellaria hastifolia | 0 |
| Trifolium repens | 1 | Galium sylvaticum | 0 | Scutellaria minor | 0 |
| Triglochin palustre | 1 | Galium truniacum | 0 | Sedum annuum | 0 |
| Tussilago farfara | 1 | Galium uliginosum | 0 | Sedum atratum | 0 |
| Valeriana officinalis | 1 | Galium valdepiosum | 0 | Sedum maximum | 0 |
| Ventenata dubia | 1 | Galium wirtgenii | 0 | Sedum villosum | 0 |
| Verbascum blattaria | 1 | Genista germanica | 0 | Selinum carvifolia | 0 |
| Verbascum thapsus | 1 | Genista pilosa | 0 | Senecio abrotanifolius | 0 |
| Veronica anagallis-aquatica | 1 | Genista sagittalis | 0 | Senecio alpinus | 0 |
| Veronica catenata | 1 | Gentiana acaulis | 0 | Senecio aquaticus | 0 |
| Veronica hederifolia | 1 | Gentiana asclepiadea | 0 | Senecio doronicum | 0 |
| Veronica serpyllifolia | 1 | Gentiana bavarica | 0 | Senecio erraticus | 0 |
| Veronica verna | 1 | Gentiana clusii | 0 | Senecio erucifolius | 0 |
| Vicia cracca | 1 | Gentiana cruciata | 0 | Senecio germanicus | 0 |
| Vicia hirsuta | 1 | Gentiana lutea | 0 | Senecio hercynicus | 0 |
| Vicia sepium | 1 | Gentiana nivalis | 0 | Senecio incanus | 0 |
| Vicia tetrasperma | 1 | Gentiana orbicularis | 0 | Senecio ovatus | 0 |

| | | | | | |
|---------------------------|---|---------------------------|---|---------------------------|---|
| Viola riviniana | 1 | Gentiana pannonica | 0 | Senecio paludosus | 0 |
| Vulpia bromoides | 1 | Gentiana pneumonanthe | 0 | Senecio rupestris | 0 |
| Aceras anthropophorum | 0 | Gentiana punctata | 0 | Senecio sarracenicus | 0 |
| Achillea atrata | 0 | Gentiana purpurea | 0 | Senecio subalpinus | 0 |
| Achillea clavennae | 0 | Gentiana utriculosa | 0 | Senecio viscosus | 0 |
| Achillea collina | 0 | Gentiana verna | 0 | Serratula tinctoria | 0 |
| Achillea macrophylla | 0 | Gentianella amarella | 0 | Seseli annuum | 0 |
| Achillea pannonica | 0 | Gentianella aspera | 0 | Seseli hippomarathrum | 0 |
| Achillea pratensis | 0 | Gentianella bohemica | 0 | Seseli libanotis | 0 |
| Achillea salicifolia | 0 | Gentianella campestris | 0 | Sesleria albicans | 0 |
| Achillea setacea | 0 | Gentianella ciliata | 0 | Sibbaldia procumbens | 0 |
| Acinos alpinus | 0 | Gentianella germanica | 0 | Silaum silaus | 0 |
| Aconitum austriacum | 0 | Gentianella lutescens | 0 | Silene armeria | 0 |
| Aconitum degenii | 0 | Gentianella tenella | 0 | Silene dioica | 0 |
| Aconitum hebegynum | 0 | Geranium bohemicum | 0 | Silene nutans | 0 |
| Aconitum lycoctonum | 0 | Geranium palustre | 0 | Silene otites | 0 |
| Aconitum pilipes | 0 | Geranium phaeum | 0 | Silene rupestris | 0 |
| Aconitum pilosiusculum | 0 | Geranium pratense | 0 | Silene viscosa | 0 |
| Aconitum tauricum | 0 | Geranium sylvaticum | 0 | Sisymbrium austriacum | 0 |
| Aconitum variegatum | 0 | Geum montanum | 0 | Sisymbrium strictissimum | 0 |
| Actaea spicata | 0 | Geum reptans | 0 | Sium latifolium | 0 |
| Adenophora liliifolia | 0 | Geum rivale | 0 | Soldanella alpicola | 0 |
| Adenostyles alliariae | 0 | Geum urbanum | 0 | Soldanella alpina | 0 |
| Adenostyles glabra | 0 | Gladiolus imbricatus | 0 | Soldanella austriaca | 0 |
| Adonis vernalis | 0 | Gladiolus palustris | 0 | Soldanella minima | 0 |
| Agrimonia eupatoria | 0 | Glaux maritima | 0 | Soldanella montana | 0 |
| Agrimonia procera | 0 | Globularia cordifolia | 0 | Solidago virgaurea | 0 |
| Agrostis agrostiflora | 0 | Globularia nudicaulis | 0 | Spergularia echinosperma | 0 |
| Agrostis alpina | 0 | Gnaphalium hoppeanum | 0 | Spergularia salina | 0 |
| Agrostis rupestris | 0 | Gnaphalium norvegicum | 0 | Spiranthes aestivalis | 0 |
| Agrostis schleicheri | 0 | Gnaphalium supinum | 0 | Spiranthes spiralis | 0 |
| Agrostis vinealis | 0 | Gnaphalium sylvaticum | 0 | Stachys alpina | 0 |
| Ajuga pyramidalis | 0 | Gnaphalium uliginosum | 0 | Stachys germanica | 0 |
| Allium angulosum | 0 | Goodyera repens | 0 | Stachys recta | 0 |
| Allium cirrhosum | 0 | Gratiola officinalis | 0 | Stachys sylvatica | 0 |
| Allium kochii | 0 | Gymnadenia conopsea | 0 | Stellaria longifolia | 0 |
| Allium lusitanicum | 0 | Gymnadenia odoratissima | 0 | Stellaria neglecta | 0 |
| Allium rotundum | 0 | Gypsophila muralis | 0 | Stellaria nemorum | 0 |
| Allium schoenoprasum | 0 | Hammarbya paludosa | 0 | Stellaria palustris | 0 |
| Allium sphaerocephalon | 0 | Hedysarum hedysaroides | 0 | Stipa borysthena | 0 |
| Allium strictum | 0 | Helianthemum alpestre | 0 | Stipa capillata | 0 |
| Allium suaveolens | 0 | Helianthemum apenninum | 0 | Stipa dasyphylla | 0 |
| Allium ursinum | 0 | Helianthemum canum | 0 | Stipa eriocalis | 0 |
| Allium victorialis | 0 | Helianthemum nummularium | 0 | Stipa pennata | 0 |
| Alopecurus rendlei | 0 | Helichrysum arenarium | 0 | Stipa pulcherrima | 0 |
| Althaea officinalis | 0 | Helictotrichon parlatorei | 0 | Stipa tirsia | 0 |
| Anacamptis pyramidalis | 0 | Helictotrichon pratense | 0 | Streptopus amplexifolius | 0 |
| Anagallis foemina | 0 | Helictotrichon pubescens | 0 | Succisa pratensis | 0 |
| Andromeda polifolia | 0 | Helictotrichon versicolor | 0 | Swertia perennis | 0 |
| Androsace lactea | 0 | Helleborus niger | 0 | Symphytum tuberosum | 0 |
| Androsace septentrionalis | 0 | Helleborus viridis | 0 | Tanacetum corymbosum | 0 |
| Anemone narcissiflora | 0 | Hepatica nobilis | 0 | Tephrosia crispa | 0 |
| Anemone nemorosa | 0 | Heracleum austriacum | 0 | Tephrosia helenitis | 0 |
| Anemone ranunculoides | 0 | Heracleum sphondylium | 0 | Tephrosia integrifolia | 0 |
| Anemone sylvestris | 0 | Hernium monorchis | 0 | Tephrosia tenuifolia | 0 |
| Angelica archangelica | 0 | Hierochloa australis | 0 | Tetragonolobus maritimus | 0 |
| Angelica palustris | 0 | Hierochloa hirta | 0 | Teucrium chamaedrys | 0 |
| Antennaria carpatia | 0 | Hierochloa odorata | 0 | Teucrium montanum | 0 |
| Anthericum liliago | 0 | Himantoglossum hircinum | 0 | Teucrium scordium | 0 |
| Anthericum ramosum | 0 | Hippocrepis comosa | 0 | Thalictrum aquilegifolium | 0 |
| Anthoxanthum alpinum | 0 | Homogyne alpina | 0 | Thalictrum flavum | 0 |
| Anthriscus nitida | 0 | Homogyne discolor | 0 | Thalictrum lucidum | 0 |

| | | | | | |
|--------------------------|---|---------------------------|---|--------------------------|---|
| Anthyllis vulneraria | 0 | Hordelymus europaeus | 0 | Thalictrum minus | 0 |
| Apera spica-venti | 0 | Horminum pyrenaicum | 0 | Thalictrum simplex | 0 |
| Apium repens | 0 | Hornungia petraea | 0 | Thesium alpinum | 0 |
| Aposyeris foetida | 0 | Hyacinthoides non-scripta | 0 | Thesium bavarum | 0 |
| Aquilegia atrata | 0 | Hydrocotyle vulgaris | 0 | Thesium ebracteatum | 0 |
| Aquilegia einseleana | 0 | Hypericum elegans | 0 | Thesium linophyllum | 0 |
| Arabis auriculata | 0 | Hypericum hirsutum | 0 | Thesium pyrenaicum | 0 |
| Arabis caerulea | 0 | Hypericum maculatum | 0 | Thesium rostratum | 0 |
| Arabis glabra | 0 | Hypericum montanum | 0 | Thymus praecox | 0 |
| Arabis hirsuta | 0 | Hypericum pulchrum | 0 | Tofieldia calyculata | 0 |
| Arabis nemorensis | 0 | Hypochaeris maculata | 0 | Tofieldia pusilla | 0 |
| Arabis pauciflora | 0 | Hypochaeris uniflora | 0 | Tolpis staticifolia | 0 |
| Arabis sagittata | 0 | Inula britannica | 0 | Tozzia alpina | 0 |
| Arabis turrita | 0 | Inula conyzae | 0 | Tragopogon orientalis | 0 |
| Arctium nemorosum | 0 | Inula germanica | 0 | Traunsteinera globosa | 0 |
| Arctostaphylos alpinus | 0 | Inula hirta | 0 | Trichophorum alpinum | 0 |
| Arctostaphylos uva-ursi | 0 | Inula salicina | 0 | Trichophorum cespitosum | 0 |
| Aremonia agrimonoides | 0 | Iris sibirica | 0 | Trientalis europaea | 0 |
| Arenaria biflora | 0 | Iris variegata | 0 | Trifolium alpestre | 0 |
| Arenaria ciliata | 0 | Isolepis setacea | 0 | Trifolium badium | 0 |
| Arnica montana | 0 | Jasione montana | 0 | Trifolium medium | 0 |
| Arnoseris minima | 0 | Juncus alpinus | 0 | Trifolium montanum | 0 |
| Artemisia laciniata | 0 | Juncus atratus | 0 | Trifolium ochroleucon | 0 |
| Artemisia scoparia | 0 | Juncus conglomeratus | 0 | Trifolium retusum | 0 |
| Artemisia umbelliformis | 0 | Juncus filiformis | 0 | Trifolium rubens | 0 |
| Arum maculatum | 0 | Juncus jacquinii | 0 | Trifolium spadiceum | 0 |
| Asarum europaeum | 0 | Juncus minutulus | 0 | Trifolium thalii | 0 |
| Asperula cynanchica | 0 | Juncus ranarius | 0 | Triglochin maritimum | 0 |
| Asperula tinctoria | 0 | Juncus sphaerocarpus | 0 | Trinia glauca | 0 |
| Aster alpinus | 0 | Juncus subnodulosus | 0 | Trisetum distichophyllum | 0 |
| Aster amellus | 0 | Juncus tenageia | 0 | Trisetum flavescens | 0 |
| Aster bellidiastrium | 0 | Juncus trifidus | 0 | Trisetum spicatum | 0 |
| Aster linosyris | 0 | Juncus triglumis | 0 | Trollius europaeus | 0 |
| Aster tripolium | 0 | Juniperus sibirica | 0 | Urtica kioviensis | 0 |
| Astragalus alpinus | 0 | Knautia dipsacifolia | 0 | Vaccinium myrtillus | 0 |
| Astragalus australis | 0 | Knautia drymeia | 0 | Vaccinium oxycoccos | 0 |
| Astragalus cicer | 0 | Knautia kitaibelii | 0 | Vaccinium uliginosum | 0 |
| Astragalus danicus | 0 | Kobresia simpliciuscula | 0 | Vaccinium vitis-idaea | 0 |
| Astragalus excapus | 0 | Koeleria glauca | 0 | Valeriana dioica | 0 |
| Astragalus frigidus | 0 | Koeleria macrantha | 0 | Valeriana montana | 0 |
| Astragalus glycyphyllos | 0 | Koeleria pyramidata | 0 | Valeriana pratensis | 0 |
| Astragalus penduliflorus | 0 | Lactuca perennis | 0 | Valeriana procurrens | 0 |
| Astrantia bavarica | 0 | Lactuca quercina | 0 | Valeriana sambucifolia | 0 |
| Astrantia major | 0 | Lactuca viminea | 0 | Valeriana saxatilis | 0 |
| Athamanta cretensis | 0 | Lappula deflexa | 0 | Valeriana supina | 0 |
| Atriplex littoralis | 0 | Laser trilobum | 0 | Valeriana versifolia | 0 |
| Atropa bella-donna | 0 | Laserpitium latifolium | 0 | Valeriana wallrothii | 0 |
| Barbarea stricta | 0 | Laserpitium prutenicum | 0 | Valerianella carinata | 0 |
| Bartsia alpina | 0 | Laserpitium siler | 0 | Valerianella dentata | 0 |
| Bassia laniflora | 0 | Lathraea squamaria | 0 | Valerianella rimosa | 0 |
| Betonica alopecuroides | 0 | Lathyrus heterophyllus | 0 | Veratrum album | 0 |
| Betonica officinalis | 0 | Lathyrus laevigatus | 0 | Verbascum densiflorum | 0 |
| Biscutella laevigata | 0 | Lathyrus linifolius | 0 | Verbascum lychnitis | 0 |
| Bistorta officinalis | 0 | Lathyrus niger | 0 | Verbascum nigrum | 0 |
| Bistorta vivipara | 0 | Lathyrus palustris | 0 | Verbascum phlomoides | 0 |
| Blackstonia acuminata | 0 | Lathyrus pannonicus | 0 | Verbascum phoeniceum | 0 |
| Blackstonia perfoliata | 0 | Lathyrus pratensis | 0 | Verbascum pulverulentum | 0 |
| Blysmus compressus | 0 | Lathyrus tuberosus | 0 | Veronica acinifolia | 0 |
| Bolboschoenus maritimus | 0 | Lathyrus vernus | 0 | Veronica alpina | 0 |
| Bolboschoenus maritimus | 0 | Lavatera thuringiaca | 0 | Veronica anagalloides | 0 |
| Bolboschoenus yagara | 0 | Leontodon helveticus | 0 | Veronica aphylla | 0 |
| Brachypodium rupestre | 0 | Leontodon incanus | 0 | Veronica bellidioides | 0 |

| | | | | | |
|--------------------------------|---|---------------------------|---|---------------------------|---|
| Briza media | 0 | Leontodon montanus | 0 | Veronica dillenii | 0 |
| Bromus benekenii | 0 | Leontopodium alpinum | 0 | Veronica fruticans | 0 |
| Bromus lepidus | 0 | Leonurus marrubiastrum | 0 | Veronica fruticulosa | 0 |
| Bromus ramosus | 0 | Lepidium graminifolium | 0 | Veronica montana | 0 |
| Buglossoides purpureoerulea | 0 | Leucanthemopsis alpina | 0 | Veronica praecox | 0 |
| Buphthalmum salicifolium | 0 | Leucanthemum adustum | 0 | Veronica scutellata | 0 |
| Bupleurum falcatum | 0 | Leucanthemum halleri | 0 | Veronica teucrium | 0 |
| Bupleurum longifolium | 0 | Leucanthemum ircutianum | 0 | Vicia cassubica | 0 |
| Bupleurum ranunculoides | 0 | Leucojum vernum | 0 | Vicia dumetorum | 0 |
| Bupleurum tenuissimum | 0 | Ligusticum mutellina | 0 | Vicia lathyroides | 0 |
| Calamagrostis arundinacea | 0 | Ligusticum mutellinoides | 0 | Vicia oroboides | 0 |
| Calamagrostis canescens | 0 | Lilium bulbiferum | 0 | Vicia pisiformis | 0 |
| Calamagrostis pseudophragmites | 0 | Lilium martagon | 0 | Vicia sylvatica | 0 |
| Calamagrostis varia | 0 | Limodorum abortivum | 0 | Vicia tenuifolia | 0 |
| Calamagrostis villosa | 0 | Limosella aquatica | 0 | Vincetoxicum hirundinaria | 0 |
| Calamintha einseleana | 0 | Linaria alpina | 0 | Viola alba | 0 |
| Calamintha menthifolia | 0 | Lindernia procumbens | 0 | Viola ambigua | 0 |
| Calamintha nepeta | 0 | Linum alpinum | 0 | Viola biflora | 0 |
| Calla palustris | 0 | Linum flavum | 0 | Viola calcarata | 0 |
| Caltha palustris | 0 | Linum perenne | 0 | Viola canina | 0 |
| Campanula alpina | 0 | Linum tenuifolium | 0 | Viola collina | 0 |
| Campanula barbata | 0 | Linum viscosum | 0 | Viola elatior | 0 |
| Campanula baumgartenii | 0 | Liparis loeselii | 0 | Viola hirta | 0 |
| Campanula bononiensis | 0 | Listera cordata | 0 | Viola kitaibeliana | 0 |
| Campanula cervicaria | 0 | Listera ovata | 0 | Viola mirabilis | 0 |
| Campanula cochleariifolia | 0 | Lithospermum officinale | 0 | Viola montana | 0 |
| Campanula glomerata | 0 | Lloydia serotina | 0 | Viola palustris | 0 |
| Campanula latifolia | 0 | Loiseleuria procumbens | 0 | Viola pumila | 0 |
| Campanula patula | 0 | Lomatogonium carinthiacum | 0 | Viola pyrenaica | 0 |
| Campanula persicifolia | 0 | Lotus tenuis | 0 | Viola reichenbachiana | 0 |
| Campanula rapunculus | 0 | Luzula alpina | 0 | Viola rupestris | 0 |
| Campanula rotundifolia | 0 | Luzula alpinopilosa | 0 | Viola schultzei | 0 |
| Campanula scheuchzeri | 0 | Luzula campestris | 0 | Viola stagnina | 0 |
| Campanula sibirica | 0 | Luzula divulgata | 0 | Viola tricolor | 0 |
| Campanula thyrsoidea | 0 | Luzula forsteri | 0 | Virga pilosa | 0 |
| Campanula trachelium | 0 | Luzula glabrata | 0 | | |
| Cardamine alpina | 0 | Luzula luzulina | 0 | | |
| Cardamine amara | 0 | Luzula multiflora | 0 | | |
| Cardamine dentata | 0 | Luzula nivea | 0 | | |
| Cardamine parviflora | 0 | Luzula pallidula | 0 | | |
| Cardamine resedifolia | 0 | Luzula pilosa | 0 | | |
| Cardamine trifolia | 0 | Luzula spicata | 0 | | |
| Cardamine udicola | 0 | Luzula sudetica | 0 | | |

APPENDIX MANUSCRIPT 2

SUPPORTING INFORMATION

Climate change will increase the naturalization risk from garden plants in Europe

Iwona Dullinger, Johannes Wessely, Oliver Bossdorf, Wayne Dawson, Franz Essl, Andreas Gattringer, Günther Klonner, Holger Kreft, Michael Kuttner, Dietmar Moser, Jan Pergl, Petr Pyšek, Wilfried Thuiller, Mark van Kleunen, Patrick Weigelt, Marten Winter, Stefan Dullinger

Appendix S1. Selected species and number of suitable cells under current and future climate.

Selected species (nomenclature as in The Plant List (<http://www.theplantlist.org/>)) and number of suitable cells under current climate and future climate (RCP2.6, RCP4.5, RCP8.5), total number of cells: 31.139.

| species | current | RCP2.6 | RCP4.5 | RCP8.5 | species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|---------------------------|---------|--------|--------|--------|--------------------------------|---------|--------|--------|--------|
| Abelmoschus manihot | 49 | 50 | 118 | 0 | Anacardium occidentale | 0 | 0 | 0 | 0 |
| Abelmoschus moschatus | 0 | 0 | 0 | 0 | Ananas comosus | 5 | 0 | 0 | 42 |
| Abrus precatorius | 0 | 0 | 4 | 6 | Angelonia angustifolia | 0 | 0 | 0 | 0 |
| Acacia pravissima | 2819 | 4577 | 3918 | 6414 | Anigozanthos flavidus | 151 | 170 | 128 | 63 |
| Acalypha hispida | 10 | 0 | 0 | 5 | Annona cherimola | 150 | 151 | 59 | 50 |
| Acalypha wilkesiana | 74 | 99 | 91 | 230 | Annona glabra | 0 | 0 | 0 | 0 |
| Acanthocereus tetragonus | 0 | 0 | 0 | 0 | Annona muricata | 0 | 0 | 0 | 0 |
| Acer buergerianum | 9230 | 12049 | 13524 | 18645 | Annona reticulata | 0 | 0 | 0 | 1 |
| Acer japonicum | 5883 | 3147 | 5336 | 1471 | Annona squamosa | 1 | 0 | 0 | 16 |
| Acer palmatum | 10247 | 15433 | 17037 | 26982 | Anoda cristata | 7954 | 11054 | 10109 | 4548 |
| Acer spicatum | 613 | 397 | 642 | 2307 | Anthurium pentaphyllum | 0 | 0 | 0 | 0 |
| Actinidia arguta | 2248 | 2317 | 3320 | 7218 | Anthurium schlechtendalii | 0 | 0 | 0 | 0 |
| Actinidia polygama | 617 | 2321 | 4178 | 2852 | Antigonon leptopus | 109 | 78 | 105 | 247 |
| Adansonia digitata | 15 | 15 | 31 | 0 | Apodytes dimidiata | 256 | 313 | 186 | 158 |
| Adenanthera pavonina | 0 | 0 | 0 | 0 | Archontophoenix cunninghamiana | 94 | 188 | 134 | 486 |
| Adenium obesum | 0 | 0 | 0 | 0 | Arctotis fastuosa | 118 | 135 | 22 | 7 |
| Aesculus glabra | 4053 | 3624 | 3537 | 8724 | Ardisia crenata | 207 | 416 | 777 | 619 |
| Agastache rugosa | 14899 | 14556 | 15299 | 16282 | Ardisia japonica | 8 | 49 | 87 | 1187 |
| Ageratina ligustrina | 0 | 0 | 3 | 24 | Aristolochia grandiflora | 0 | 0 | 0 | 0 |
| Ageratum conyzoides | 126 | 147 | 148 | 269 | Artemisia ludoviciana | 2002 | 2337 | 2026 | 245 |
| Albizia chinensis | 125 | 193 | 143 | 259 | Artocarpus altilis | 0 | 0 | 0 | 0 |
| Albizia lebeck | 175 | 391 | 479 | 863 | Artocarpus heterophyllus | 133 | 323 | 254 | 690 |
| Albizia saman | 0 | 0 | 0 | 0 | Arum palaestinum | 24 | 16 | 5 | 1 |
| Albica bracteata | 2107 | 996 | 568 | 1423 | Arundina graminifolia | 120 | 274 | 239 | 282 |
| Albica canadensis | 117 | 87 | 15 | 1 | Asimina triloba | 2623 | 2589 | 3901 | 1 |
| Aleurites moluccana | 155 | 150 | 144 | 290 | Asparagus africanus | 358 | 434 | 195 | 5 |
| Allamanda cathartica | 0 | 0 | 0 | 0 | Asparagus declinatus | 147 | 171 | 78 | 225 |
| Allium tuberosum | 15835 | 16953 | 18100 | 19784 | Asparagus falcatus | 75 | 154 | 45 | 73 |
| Allocauarina littoralis | 389 | 917 | 724 | 684 | Asparagus retrofractus | 23 | 38 | 4 | 94 |
| Allocauarina verticillata | 1214 | 2591 | 2380 | 2524 | Asparagus scandens | 712 | 1344 | 812 | 1534 |
| Alnus nepalensis | 158 | 250 | 352 | 175 | Asparagus virgatus | 564 | 589 | 461 | 8 |
| Alocasia macrorrhizos | 45 | 92 | 132 | 279 | Atriplex canescens | 2612 | 2484 | 2595 | 7 |
| Alstonia scholaris | 0 | 0 | 12 | 15 | Atriplex nummularia | 2415 | 2583 | 1677 | 741 |
| Alternanthera ficoidea | 0 | 0 | 0 | 188 | Averrhoa carambola | 0 | 0 | 0 | 0 |
| Ammobium alatum | 5089 | 6134 | 5468 | 1079 | Banisteriopsis caapi | 0 | 0 | 0 | 0 |
| Ampelopsis glandulosa | 92 | 205 | 258 | 760 | Banksia ericifolia | 368 | 643 | 700 | 971 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|----------------------------|---------|--------|--------|--------|
| Barleria cristata | 437 | 617 | 644 | 136 |
| Barringtonia asiatica | 0 | 0 | 0 | 0 |
| Bartlettina sordida | 88 | 160 | 47 | 173 |
| Basella alba | 501 | 881 | 1222 | 1408 |
| Bauhinia galpinii | 225 | 242 | 180 | 33 |
| Bauhinia purpurea | 45 | 136 | 144 | 381 |
| Bauhinia variegata | 1361 | 1772 | 1567 | 1925 |
| Begonia cucullata | 330 | 339 | 305 | 1443 |
| Begonia heracleifolia | 0 | 0 | 0 | 0 |
| Begonia humilis | 0 | 0 | 0 | 0 |
| Begonia nelumbiifolia | 0 | 0 | 0 | 0 |
| Berberis glaucocarpa | 1854 | 1781 | 1943 | 5800 |
| Berberis repens | 3618 | 2810 | 1371 | 0 |
| Bertholletia excelsa | 0 | 0 | 0 | 0 |
| Betula platyphylla | 5158 | 6116 | 6491 | 1678 |
| Bignonia capreolata | 33 | 194 | 267 | 5979 |
| Billardiera heterophylla | 1192 | 1534 | 1460 | 261 |
| Bixa orellana | 0 | 0 | 0 | 0 |
| Bocconia frutescens | 13 | 10 | 0 | 0 |
| Boltonia asteroides | 375 | 609 | 577 | 1669 |
| Bomarea multiflora | 237 | 498 | 66 | 187 |
| Bombax ceiba | 0 | 10 | 20 | 5 |
| Bouteloua curtipendula | 1266 | 2120 | 2238 | 2893 |
| Brachychiton acerifolius | 1153 | 2432 | 2632 | 4449 |
| Brachychiton discolor | 2289 | 3811 | 4059 | 4380 |
| Brachychiton populneus | 2283 | 3375 | 2933 | 1937 |
| Brexia madagascariensis | 0 | 0 | 0 | 0 |
| Breynia disticha | 0 | 1 | 0 | 8 |
| Bromus briziformis | 6877 | 3241 | 2186 | 0 |
| Bromus danthoniae | 1306 | 548 | 1035 | 0 |
| Browallia americana | 35 | 0 | 0 | 0 |
| Brownea coccinea | 0 | 0 | 0 | 0 |
| Brownea grandiceps | 0 | 0 | 0 | 0 |
| Brugmansia sanguinea | 3 | 36 | 0 | 205 |
| Brunfelsia uniflora | 238 | 368 | 398 | 731 |
| Bucida buceras | 0 | 0 | 0 | 0 |
| Buddleja asiatica | 836 | 1326 | 1510 | 2027 |
| Buddleja indica | 0 | 0 | 0 | 0 |
| Buddleja saligna | 403 | 588 | 190 | 20 |
| Buddleja salviifolia | 309 | 638 | 251 | 0 |
| Buddleja stachyoides | 257 | 374 | 545 | 790 |
| Bulbine semibarbata | 3206 | 4754 | 4163 | 1153 |
| Bursera simaruba | 0 | 0 | 1 | 1 |
| Caesalpinia coriaria | 0 | 0 | 0 | 0 |
| Caesalpinia pulcherrima | 0 | 0 | 46 | 48 |
| Caladium bicolor | 0 | 1 | 0 | 11 |
| Calceolaria chelidonioides | 2572 | 2985 | 2318 | 2450 |
| Calliandra haematocephala | 3 | 18 | 32 | 125 |
| Calliandra houstoniana | 0 | 0 | 0 | 0 |
| Calliandra surinamensis | 0 | 0 | 0 | 0 |
| Callicarpa dichotoma | 284 | 492 | 1013 | 581 |
| Callicarpa japonica | 233 | 392 | 772 | 1503 |
| Callisia repens | 230 | 339 | 366 | 466 |
| Callistemon speciosus | 658 | 814 | 514 | 507 |
| Callistemon viminalis | 1182 | 1226 | 752 | 468 |
| Calophyllum inophyllum | 0 | 0 | 0 | 0 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|--------------------------|---------|--------|--------|--------|
| Calotropis procera | 9 | 246 | 609 | 832 |
| Calystegia hederacea | 62 | 120 | 154 | 2 |
| Camellia japonica | 1914 | 2790 | 2971 | 4352 |
| Camellia sinensis | 456 | 767 | 1195 | 1791 |
| Campanula punctata | 1268 | 2895 | 3221 | 2597 |
| Cananga odorata | 0 | 0 | 0 | 0 |
| Canavalia cathartica | 223 | 390 | 253 | 769 |
| Canavalia ensiformis | 170 | 175 | 150 | 468 |
| Canna glauca | 78 | 58 | 57 | 12 |
| Cardiocrinum cordatum | 25 | 52 | 40 | 0 |
| Carica papaya | 7 | 2 | 26 | 27 |
| Carissa macrocarpa | 1096 | 1304 | 1536 | 1994 |
| Carpinus caroliniana | 1229 | 2034 | 2059 | 5229 |
| Carya illinoensis | 3738 | 6716 | 8203 | 7597 |
| Cascabela thevetia | 4 | 4 | 4 | 2 |
| Cassia fistula | 0 | 0 | 0 | 0 |
| Cassia grandis | 0 | 0 | 0 | 0 |
| Castanea mollissima | 1897 | 2725 | 3524 | 3126 |
| Castanospermum australe | 60 | 144 | 133 | 248 |
| Casuarina cunninghamiana | 2074 | 2719 | 2421 | 2164 |
| Casuarina glauca | 117 | 330 | 303 | 654 |
| Cedrela odorata | 23 | 13 | 10 | 28 |
| Ceiba pentandra | 0 | 0 | 0 | 0 |
| Celastrus scandens | 661 | 1424 | 1164 | 6957 |
| Centrosema virginianum | 267 | 841 | 981 | 3956 |
| Ceratopetalum gummiferum | 69 | 217 | 142 | 0 |
| Ceratotheca triloba | 176 | 166 | 120 | 60 |
| Cercestis mirabilis | 0 | 0 | 0 | 0 |
| Cercidiphyllum japonicum | 13777 | 14119 | 13621 | 20671 |
| Cestrum aurantiacum | 1406 | 1917 | 1633 | 1880 |
| Cestrum fasciculatum | 578 | 916 | 815 | 132 |
| Cestrum nocturnum | 134 | 233 | 234 | 657 |
| Chamaedorea elegans | 0 | 34 | 13 | 1 |
| Chamelaucium uncinatum | 353 | 459 | 292 | 767 |
| Cheilocostus speciosus | 0 | 0 | 0 | 0 |
| Chenopodium quinoa | 5673 | 5588 | 4848 | 7331 |
| Chrysobalanus icaco | 0 | 0 | 0 | 0 |
| Chrysophyllum cainito | 0 | 0 | 0 | 0 |
| Chrysophyllum oliviforme | 0 | 0 | 0 | 0 |
| Chrysothemis pulchella | 0 | 0 | 0 | 0 |
| Cinnamomum camphora | 3224 | 6532 | 9533 | 20213 |
| Cissus alata | 0 | 0 | 0 | 0 |
| Cissus antarctica | 211 | 472 | 489 | 540 |
| Cissus quadrangularis | 0 | 0 | 0 | 1 |
| Cissus rotundifolia | 26 | 18 | 9 | 33 |
| Cissus verticillata | 0 | 0 | 0 | 11 |
| Citharexylum spinosum | 0 | 0 | 0 | 0 |
| Citrus aurantiifolia | 0 | 0 | 0 | 0 |
| Citrus maxima | 553 | 924 | 1365 | 1368 |
| Clarkia amoena | 10995 | 11548 | 6560 | 502 |
| Clarkia pulchella | 14766 | 13809 | 9218 | 4038 |
| Clematis paniculata | 589 | 663 | 699 | 2447 |
| Clematis tangutica | 6472 | 6728 | 2791 | 433 |
| Clematis terniflora | 2993 | 3817 | 4528 | 12846 |
| Cleome gynandra | 867 | 723 | 510 | 593 |
| Clerodendrum bungei | 1109 | 1306 | 1365 | 77 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|---------------------------|---------|--------|--------|--------|
| Clerodendrum chinense | 24 | 100 | 155 | 236 |
| Clerodendrum splendens | 0 | 0 | 0 | 0 |
| Clerodendrum trichotomum | 1872 | 2104 | 2926 | 5104 |
| Cleyera japonica | 71 | 170 | 573 | 1025 |
| Clidemia hirta | 0 | 0 | 0 | 0 |
| Clitoria ternatea | 0 | 0 | 0 | 19 |
| Clusia rosea | 0 | 0 | 0 | 0 |
| Cobaea scandens | 895 | 1316 | 774 | 796 |
| Coccinia grandis | 0 | 0 | 0 | 0 |
| Coccoloba uvifera | 0 | 0 | 0 | 1 |
| Cochlospermum vitifolium | 0 | 0 | 0 | 0 |
| Cocos nucifera | 0 | 0 | 0 | 0 |
| Codiaeum variegatum | 0 | 0 | 0 | 0 |
| Coffea liberica | 0 | 0 | 0 | 0 |
| Cola acuminata | 0 | 0 | 0 | 0 |
| Combretum grandiflorum | 0 | 0 | 0 | 0 |
| Combretum indicum | 0 | 0 | 0 | 0 |
| Commelina benghalensis | 924 | 957 | 606 | 686 |
| Commelina coelestis | 26 | 29 | 107 | 102 |
| Conicosia pugioniformis | 200 | 525 | 94 | 13 |
| Coprosma robusta | 103 | 278 | 335 | 213 |
| Corchorus olitorius | 523 | 860 | 1358 | 48 |
| Cordyline fruticosa | 0 | 0 | 0 | 11 |
| Cortaderia jubata | 2630 | 3624 | 2687 | 3213 |
| Corylus heterophylla | 361 | 547 | 709 | 358 |
| Corymbia citriodora | 978 | 1310 | 1115 | 2091 |
| Cosmos sulphureus | 340 | 786 | 790 | 52 |
| Cotoneaster glaucophyllus | 4511 | 4857 | 4226 | 3659 |
| Couroupita guianensis | 0 | 0 | 0 | 0 |
| Crassula ericoides | 28 | 121 | 53 | 0 |
| Crataegus pubescens | 0 | 8 | 2 | 10 |
| Crescentia cujete | 0 | 0 | 0 | 0 |
| Crinum asiaticum | 396 | 1019 | 1151 | 3489 |
| Crinum zeylanicum | 0 | 0 | 0 | 0 |
| Crotalaria capensis | 239 | 302 | 179 | 3 |
| Crotalaria juncea | 272 | 399 | 414 | 832 |
| Crotalaria micans | 275 | 495 | 457 | 693 |
| Crotalaria retusa | 0 | 0 | 0 | 0 |
| Cucumis anguria | 246 | 232 | 208 | 137 |
| Cucumis dipsaceus | 69 | 69 | 46 | 81 |
| Cucumis metuliferus | 169 | 218 | 131 | 33 |
| Cucurbita argyrosperma | 16 | 15 | 0 | 0 |
| Cucurbita ficifolia | 6296 | 8862 | 7351 | 7402 |
| Cucurbita moschata | 3914 | 4257 | 3412 | 695 |
| Cuphea hyssopifolia | 117 | 156 | 142 | 448 |
| Curcuma longa | 24 | 19 | 65 | 248 |
| Cymbopogon nardus | 636 | 1027 | 849 | 463 |
| Cynoglossum amabile | 14342 | 16365 | 12337 | 13374 |
| Cynoglossum zeylanicum | 3356 | 3819 | 3675 | 4026 |
| Cyperus albostratus | 937 | 1864 | 1420 | 20 |
| Cytisus proliferus | 5877 | 6617 | 5373 | 6264 |
| Dahlia imperialis | 0 | 4 | 0 | 0 |
| Dahlia pinnata | 14781 | 18785 | 15559 | 18081 |
| Datura ceratocaula | 44 | 23 | 1 | 182 |
| Debregeasia longifolia | 124 | 287 | 371 | 463 |
| Delonix regia | 0 | 0 | 7 | 6 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|-----------------------------|---------|--------|--------|--------|
| Desmodium elegans | 894 | 1148 | 1111 | 1 |
| Deutzia crenata | 4055 | 4519 | 5678 | 187 |
| Deutzia gracilis | 3269 | 2718 | 4135 | 569 |
| Dianella ensifolia | 429 | 811 | 937 | 1059 |
| Dieffenbachia seguine | 0 | 0 | 0 | 0 |
| Dimorphotheca cuneata | 0 | 0 | 0 | 0 |
| Dimorphotheca pluvialis | 7421 | 9466 | 6722 | 6399 |
| Dimorphotheca sinuata | 3556 | 3421 | 3405 | 13 |
| Dioscorea bulbifera | 36 | 59 | 114 | 107 |
| Dioscorea mexicana | 0 | 0 | 0 | 0 |
| Dioscorea polystachya | 493 | 1036 | 1426 | 4064 |
| Diospyros virginiana | 252 | 1043 | 1446 | 4067 |
| Diplocyclos palmatus | 218 | 236 | 241 | 146 |
| Dodonaea viscosa | 3476 | 4875 | 4281 | 4275 |
| Dolichandra unguis-cati | 348 | 374 | 330 | 469 |
| Dombeya burgessiae | 137 | 120 | 78 | 34 |
| Dombeya tiliacea | 0 | 0 | 15 | 0 |
| Dorotheanthus bellidiformis | 5 | 19 | 3 | 3 |
| Dorstenia contrajerva | 0 | 0 | 0 | 0 |
| Dracaena fragrans | 3 | 3 | 0 | 6 |
| Dracaena reflexa | 0 | 0 | 0 | 0 |
| Drosera aliciae | 1911 | 2879 | 3067 | 4193 |
| Drosera binata | 2555 | 3724 | 3761 | 4535 |
| Drosera capillaris | 16 | 53 | 115 | 189 |
| Echeveria secunda | 0 | 0 | 0 | 0 |
| Echinochloa polystachya | 100 | 146 | 92 | 46 |
| Echinodorus cordifolius | 166 | 1115 | 1967 | 3801 |
| Echinodorus subalatus | 0 | 0 | 0 | 0 |
| Ehretia acuminata | 1243 | 2769 | 3075 | 2931 |
| Eichhornia azurea | 0 | 0 | 0 | 0 |
| Elaeis guineensis | 0 | 0 | 0 | 0 |
| Embothrium coccineum | 1920 | 2165 | 1384 | 603 |
| Emilia sonchifolia | 57 | 84 | 116 | 212 |
| Empetrum rubrum | 438 | 366 | 116 | 618 |
| Entada phaseoloides | 0 | 0 | 7 | 24 |
| Epacris impressa | 1195 | 1978 | 1342 | 676 |
| Epipremnum pinnatum | 1 | 0 | 0 | 5 |
| Eragrostis trichodes | 152 | 287 | 519 | 28 |
| Erica glandulosa | 34 | 76 | 31 | 0 |
| Eriochloa villosa | 201 | 228 | 329 | 727 |
| Erythrina crista-galli | 998 | 1480 | 1588 | 3186 |
| Erythrina herbacea | 1 | 107 | 42 | 1744 |
| Etlingera elatior | 0 | 6 | 7 | 151 |
| Eucalyptus cinerea | 1792 | 2585 | 2708 | 3272 |
| Eucalyptus cladocalyx | 1596 | 1407 | 656 | 733 |
| Eucalyptus cornuta | 290 | 295 | 145 | 0 |
| Eucalyptus dalrympleana | 767 | 1329 | 1538 | 886 |
| Eucalyptus goniocalyx | 1021 | 1722 | 1169 | 93 |
| Eucalyptus leucoxydon | 1015 | 1473 | 975 | 985 |
| Eucalyptus nitens | 915 | 1514 | 1767 | 731 |
| Eucalyptus ovata | 1973 | 2577 | 1942 | 454 |
| Eucalyptus rubida | 485 | 475 | 267 | 0 |
| Eugenia uniflora | 209 | 228 | 219 | 506 |
| Eulophia alta | 0 | 0 | 0 | 0 |
| Euonymus alatus | 2569 | 2286 | 3995 | 967 |
| Euonymus hamiltonianus | 5227 | 7152 | 6647 | 47 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|--------------------------|---------|--------|--------|--------|
| Euphorbia balsamifera | 0 | 1 | 0 | 0 |
| Euphorbia leucocephala | 0 | 0 | 0 | 0 |
| Euphorbia mauritanica | 109 | 145 | 32 | 18 |
| Euphorbia tithymaloides | 22 | 15 | 25 | 135 |
| Euryops abrotanifolius | 201 | 465 | 165 | 403 |
| Euryops chrysanthemoides | 21 | 12 | 0 | 0 |
| Ficus auriculata | 58 | 60 | 90 | 3 |
| Ficus benjamina | 74 | 84 | 91 | 110 |
| Ficus drupacea | 0 | 0 | 0 | 55 |
| Ficus elastica | 2331 | 2760 | 2490 | 3563 |
| Ficus erecta | 100 | 309 | 285 | 700 |
| Ficus lutea | 0 | 0 | 0 | 0 |
| Ficus palmata | 2756 | 2948 | 3191 | 636 |
| Ficus racemosa | 0 | 0 | 21 | 0 |
| Ficus virens | 77 | 59 | 65 | 75 |
| Firmiana simplex | 2121 | 2779 | 2815 | 6734 |
| Fragaria chiloensis | 6812 | 8394 | 6067 | 8576 |
| Frangula purshiana | 568 | 200 | 232 | 8 |
| Fraxinus nigra | 68 | 229 | 257 | 2809 |
| Fraxinus uhdei | 0 | 0 | 1 | 40 |
| Freesia laxa | 666 | 915 | 1097 | 791 |
| Fuchsia paniculata | 0 | 3 | 0 | 6 |
| Galphimia glauca | 8 | 8 | 64 | 122 |
| Galphimia gracilis | 0 | 0 | 2 | 0 |
| Garcinia livingstonei | 43 | 34 | 31 | 0 |
| Gardenia jasminoides | 521 | 1005 | 1301 | 2326 |
| Geitonoplesium cymosum | 395 | 716 | 767 | 197 |
| Geranium incanum | 739 | 1106 | 510 | 0 |
| Geranium thunbergii | 3044 | 5675 | 7898 | 16788 |
| Gerbera jamesonii | 487 | 501 | 722 | 616 |
| Gibasis pellucida | 64 | 215 | 161 | 440 |
| Gilia tricolor | 1641 | 799 | 398 | 141 |
| Gladiolus papilio | 351 | 508 | 430 | 0 |
| Gladiolus tristis | 866 | 1415 | 956 | 386 |
| Gladiolus undulatus | 1173 | 1931 | 1726 | 1456 |
| Glandularia peruviana | 704 | 972 | 1115 | 2591 |
| Glandularia tenera | 4094 | 5932 | 5934 | 7734 |
| Gloriosa superba | 103 | 93 | 57 | 10 |
| Gloxinia perennis | 0 | 0 | 0 | 0 |
| Gloxinia sylvatica | 0 | 0 | 0 | 0 |
| Gmelina arborea | 0 | 0 | 0 | 0 |
| Gomphrena globosa | 1229 | 1903 | 2107 | 1738 |
| Grevillea banksii | 271 | 461 | 302 | 158 |
| Grevillea juniperina | 1002 | 1822 | 1282 | 796 |
| Grevillea rosmarinifolia | 1505 | 2094 | 892 | 110 |
| Hakea eriantha | 330 | 509 | 441 | 292 |
| Hakea laurina | 397 | 478 | 233 | 520 |
| Haloragis erecta | 1010 | 1380 | 1148 | 2517 |
| Hamelia patens | 7 | 6 | 5 | 15 |
| Hardenbergia comptoniana | 123 | 178 | 133 | 159 |
| Harpephyllum caffrum | 148 | 138 | 74 | 61 |
| Harrisia pomanensis | 354 | 333 | 259 | 451 |
| Hebenstretia dentata | 402 | 756 | 508 | 9 |
| Hedychium coronarium | 429 | 831 | 775 | 881 |
| Heimia salicifolia | 156 | 391 | 348 | 760 |
| Helanthium bolivianum | 0 | 0 | 0 | 0 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|--------------------------------|---------|--------|--------|--------|
| Helanthium tenellum | 37 | 123 | 258 | 41 |
| Helenium bigelovii | 747 | 339 | 272 | 0 |
| Helianthus angustifolius | 775 | 1332 | 1716 | 13329 |
| Helianthus debilis | 11600 | 17402 | 17524 | 25377 |
| Helianthus giganteus | 13563 | 13499 | 14464 | 15880 |
| Helianthus salicifolius | 13 | 0 | 6 | 0 |
| Herbertia lahue | 462 | 1795 | 1830 | 9794 |
| Hesperantha coccinea | 2844 | 2721 | 2309 | 436 |
| Hesperantha falcata | 149 | 271 | 83 | 79 |
| Heterocentron subtriplinervium | 0 | 1 | 0 | 0 |
| Heterotis rotundifolia | 0 | 0 | 0 | 0 |
| Hevea brasiliensis | 0 | 0 | 0 | 0 |
| Hibiscus acetosella | 0 | 0 | 0 | 0 |
| Hibiscus diversifolius | 225 | 340 | 242 | 540 |
| Hibiscus mutabilis | 245 | 381 | 729 | 1148 |
| Hibiscus sabdariffa | 0 | 0 | 0 | 0 |
| Hibiscus schizopetalus | 0 | 0 | 0 | 34 |
| Hibiscus tiliaceus | 0 | 0 | 0 | 24 |
| Hippeastrum puniceum | 0 | 0 | 0 | 0 |
| Hippobroma longiflora | 0 | 0 | 0 | 0 |
| Hiptage benghalensis | 0 | 15 | 28 | 0 |
| Holmskioldia sanguinea | 126 | 245 | 288 | 7 |
| Homalanthus populifolius | 982 | 2284 | 2824 | 4206 |
| Homalocladium platycladum | 15 | 15 | 50 | 199 |
| Houstonia caerulea | 402 | 392 | 413 | 549 |
| Houttuynia cordata | 6902 | 9648 | 11545 | 23910 |
| Hovea pungens | 195 | 121 | 24 | 5 |
| Hovenia dulcis | 749 | 1098 | 1397 | 592 |
| Hoya australis | 74 | 90 | 98 | 157 |
| Hoya carnosa | 4376 | 5303 | 6490 | 11799 |
| Hura crepitans | 0 | 0 | 0 | 0 |
| Hydrangea paniculata | 4176 | 4430 | 5366 | 7561 |
| Hydrocleys nymphoides | 197 | 543 | 607 | 261 |
| Hydrocotyle americana | 517 | 385 | 201 | 6 |
| Hydrolea spinosa | 1 | 9 | 20 | 2 |
| Hylotelephium erythrostictum | 4818 | 4137 | 3512 | 3447 |
| Hymenocallis littoralis | 0 | 0 | 0 | 0 |
| Hypericum gramineum | 2433 | 3239 | 2987 | 2408 |
| Hypericum hypericoides | 606 | 1503 | 1828 | 3485 |
| Hypericum patulum | 8804 | 10298 | 12032 | 17196 |
| Hypoestes aristata | 249 | 318 | 315 | 1 |
| Hypoestes phyllostachya | 22 | 17 | 25 | 7 |
| Idesia polycarpa | 718 | 946 | 1468 | 1893 |
| Ilex crenata | 1437 | 2333 | 3049 | 4906 |
| Ilex paraguayensis | 190 | 197 | 165 | 168 |
| Ilex rotunda | 95 | 121 | 398 | 841 |
| Indigofera tinctoria | 779 | 788 | 1059 | 180 |
| Inga edulis | 5 | 9 | 7 | 20 |
| Ipomoea alba | 76 | 70 | 83 | 209 |
| Ipomoea cairica | 1228 | 1566 | 1214 | 2923 |
| Ipomoea carnea | 137 | 118 | 103 | 211 |
| Ipomoea pes-caprae | 2 | 4 | 0 | 624 |
| Ipomoea tricolor | 678 | 638 | 709 | 192 |
| Iris domestica | 3131 | 4694 | 5976 | 6297 |
| Iris japonica | 4156 | 8009 | 10357 | 12968 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|-----------------------------|---------|--------|--------|--------|
| Iris sanguinea | 2164 | 2475 | 347 | 1863 |
| Isotoma fluviatilis | 1005 | 1663 | 1280 | 2500 |
| Ixia polystachya | 383 | 813 | 590 | 697 |
| Ixora coccinea | 0 | 0 | 0 | 0 |
| Jasminum dichotomum | 0 | 0 | 0 | 0 |
| Jasminum fluminense | 163 | 145 | 80 | 62 |
| Jasminum grandiflorum | 3033 | 3543 | 3613 | 3804 |
| Jasminum multiflorum | 0 | 0 | 0 | 0 |
| Jasminum polyanthum | 2026 | 3078 | 3005 | 4055 |
| Jasminum simplicifolium | 346 | 764 | 633 | 2002 |
| Justicia betonica | 186 | 211 | 169 | 192 |
| Justicia brandegeana | 353 | 329 | 212 | 178 |
| Justicia spicigera | 0 | 0 | 0 | 0 |
| Kalanchoe crenata | 37 | 34 | 25 | 1 |
| Kalopanax septemlobus | 517 | 397 | 601 | 826 |
| Kennedia rubicunda | 266 | 514 | 479 | 532 |
| Kigelia africana | 102 | 81 | 67 | 51 |
| Kolkwitzia amabilis | 14668 | 16051 | 13306 | 212 |
| Kummerowia stipulacea | 176 | 901 | 728 | 2011 |
| Kummerowia striata | 679 | 1768 | 1792 | 4134 |
| Kunzea ambigua | 1018 | 2112 | 1841 | 2542 |
| Kunzea ericoides | 1008 | 1158 | 831 | 182 |
| Lablab purpureus | 1007 | 1639 | 1266 | 1617 |
| Laelia rubescens | 0 | 0 | 0 | 0 |
| Lagerstroemia speciosa | 0 | 0 | 0 | 0 |
| Lampranthus spectabilis | 1945 | 3370 | 3215 | 3874 |
| Lawsonia inermis | 0 | 0 | 0 | 2 |
| Leonotis leonurus | 736 | 1032 | 565 | 635 |
| Leonotis ocyimifolia | 284 | 567 | 212 | 6 |
| Leonurus japonicus | 954 | 1496 | 1591 | 1359 |
| Leonurus sibiricus | 510 | 672 | 723 | 1552 |
| Leptospermum polygalifolium | 425 | 761 | 723 | 914 |
| Lespedeza bicolor | 985 | 1764 | 2251 | 2655 |
| Lespedeza cyrtobotrya | 8 | 23 | 42 | 0 |
| Lespedeza thunbergii | 1721 | 1901 | 2572 | 333 |
| Leucophyllum frutescens | 72 | 359 | 471 | 680 |
| Ligustrum obtusifolium | 1638 | 2872 | 3215 | 10229 |
| Ligustrum tschonoskii | 198 | 297 | 615 | 830 |
| Lilium formosanum | 1528 | 2761 | 4176 | 8288 |
| Limnium laevigatum | 399 | 465 | 503 | 250 |
| Limncharis flava | 0 | 0 | 0 | 211 |
| Linaria maroccana | 17489 | 18747 | 14919 | 16882 |
| Linum grandiflorum | 15546 | 15368 | 12440 | 1749 |
| Liriope muscari | 1019 | 2473 | 5180 | 4086 |
| Liriope spicata | 282 | 556 | 1033 | 2983 |
| Livistona australis | 152 | 244 | 241 | 458 |
| Lobelia cardinalis | 1675 | 3783 | 3770 | 9602 |
| Lobelia inflata | 3485 | 2077 | 2782 | 191 |
| Lonicera sempervirens | 1562 | 2310 | 3011 | 23 |
| Lophostemon confertus | 321 | 548 | 759 | 1462 |
| Ludwigia alternifolia | 0 | 104 | 391 | 995 |
| Ludwigia octovalvis | 114 | 145 | 121 | 1128 |
| Ludwigia peruviana | 492 | 822 | 743 | 1432 |
| Luffa cylindrica | 0 | 0 | 0 | 2 |
| Luma apiculata | 2599 | 3218 | 2874 | 3333 |
| Lupinus mexicanus | 0 | 0 | 0 | 0 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|---------------------------|---------|--------|--------|--------|
| Lycianthes rantonnetii | 3831 | 5168 | 5101 | 8628 |
| Lycoris radiata | 175 | 389 | 369 | 2043 |
| Lysimachia japonica | 213 | 199 | 459 | 113 |
| Maackia amurensis | 2886 | 4501 | 6737 | 11332 |
| Magnolia grandiflora | 16776 | 21678 | 22127 | 27286 |
| Magnolia kobus | 5249 | 6592 | 4472 | 16495 |
| Magnolia obovata | 834 | 2064 | 2539 | 15253 |
| Malephora crocea | 1061 | 1337 | 459 | 71 |
| Mallotus philippensis | 226 | 358 | 539 | 689 |
| Malpighia emarginata | 0 | 0 | 0 | 1 |
| Malpighia glabra | 0 | 0 | 0 | 0 |
| Malpighia mexicana | 0 | 0 | 0 | 0 |
| Malus prunifolia | 8835 | 4370 | 5756 | 34 |
| Malva assurgentiflora | 92 | 186 | 61 | 0 |
| Malvaviscus arboreus | 0 | 0 | 0 | 0 |
| Mammea americana | 0 | 0 | 0 | 0 |
| Mandevilla laxa | 209 | 283 | 427 | 89 |
| Mangifera indica | 93 | 89 | 83 | 118 |
| Manilkara zapota | 0 | 0 | 0 | 0 |
| Maranta arundinacea | 0 | 0 | 0 | 0 |
| Margyricarpus pinnatus | 2210 | 2873 | 2256 | 2458 |
| Martynia annua | 0 | 9 | 1 | 0 |
| Maurandya antirrhiniflora | 2865 | 3151 | 3632 | 583 |
| Mazus pumilus | 1908 | 3469 | 4573 | 5641 |
| Melaleuca hypericifolia | 558 | 1152 | 948 | 1179 |
| Melastoma malabathricum | 191 | 247 | 340 | 334 |
| Melanthus major | 759 | 1505 | 818 | 399 |
| Melinis repens | 1616 | 1670 | 1091 | 1090 |
| Melothria pendula | 19 | 76 | 302 | 1491 |
| Miconia calvescens | 1 | 1 | 10 | 6 |
| Micranthemum umbrosum | 142 | 375 | 617 | 1806 |
| Mimosa pigra | 0 | 0 | 0 | 1 |
| Mimusops elengi | 0 | 0 | 0 | 0 |
| Molineria capitulata | 32 | 40 | 194 | 392 |
| Momordica balsamina | 3521 | 3939 | 3323 | 3736 |
| Momordica charantia | 10 | 15 | 25 | 55 |
| Momordica cochinchinensis | 9 | 43 | 59 | 156 |
| Monarda fistulosa | 9327 | 9366 | 9320 | 12951 |
| Monarda punctata | 752 | 3234 | 4081 | 8728 |
| Monochoria vaginalis | 108 | 155 | 192 | 245 |
| Moraea flaccida | 1054 | 2016 | 1859 | 2728 |
| Moraea fugax | 537 | 621 | 231 | 9 |
| Moraea miniata | 409 | 868 | 421 | 344 |
| Moraea polystachya | 64 | 76 | 15 | 2 |
| Moraea setifolia | 1036 | 1839 | 1135 | 113 |
| Morinda citrifolia | 0 | 0 | 0 | 0 |
| Moringa oleifera | 5 | 1 | 9 | 29 |
| Mucuna pruriens | 5 | 3 | 4 | 1 |
| Muehlenbeckia axillaris | 498 | 799 | 1000 | 1355 |
| Mukia maderaspatana | 165 | 195 | 215 | 458 |
| Musa acuminata | 6840 | 8413 | 8120 | 11095 |
| Myrica rubra | 255 | 410 | 771 | 1167 |
| Myrmecophila tibicinis | 0 | 0 | 0 | 0 |
| Nandina domestica | 6306 | 10941 | 15084 | 23006 |
| Nepeta racemosa | 14417 | 14927 | 11644 | 4787 |
| Neptunia oleracea | 0 | 0 | 0 | 0 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|---------------------------|---------|--------|--------|--------|
| Neptunia plena | 0 | 0 | 0 | 0 |
| Nertera granadensis | 1564 | 1757 | 1332 | 1529 |
| Nicotiana acuminata | 2053 | 1377 | 771 | 3 |
| Nicotiana longiflora | 4951 | 6196 | 6191 | 10607 |
| Nicotiana sylvestris | 10675 | 12394 | 10947 | 19704 |
| Nothoscordum bivalve | 399 | 1678 | 2331 | 3805 |
| Nymphaea nouchali | 298 | 407 | 265 | 46 |
| Nymphaea odorata | 5889 | 6963 | 8164 | 19385 |
| Ochroma pyramidale | 0 | 0 | 0 | 0 |
| Ocimum americanum | 710 | 530 | 313 | 273 |
| Ocimum gratissimum | 228 | 228 | 180 | 252 |
| Ocimum | | | | |
| kilimandscharicum | 24 | 70 | 76 | 177 |
| Ocimum tenuiflorum | 0 | 0 | 0 | 184 |
| Odontonema tubaeforme | 0 | 2 | 2 | 9 |
| Oeceoclades maculata | 32 | 26 | 44 | 34 |
| Oenothera drummondii | 2140 | 2442 | 3226 | 5382 |
| Oenothera perennis | 4209 | 3138 | 4757 | 3303 |
| Opuntia aurantiaca | 197 | 890 | 596 | 8 |
| Opuntia basilaris | 2517 | 2157 | 2246 | 0 |
| Opuntia humifusa | 1454 | 4830 | 5307 | 10246 |
| Opuntia leucotricha | 9 | 17 | 6 | 0 |
| Opuntia polyacantha | 659 | 773 | 806 | 210 |
| Ornithogalum thyrsoides | 497 | 777 | 290 | 1117 |
| Orthrosanthus | | | | |
| chimboraensis | 147 | 403 | 216 | 142 |
| Osmanthus heterophyllus | 1364 | 2171 | 3077 | 5391 |
| Osteospermum ecklonis | 6422 | 9774 | 6946 | 4810 |
| Oxalis depressa | 1497 | 1418 | 854 | 0 |
| Oxalis spiralis | 589 | 1166 | 751 | 930 |
| Oxalis tuberosa | 0 | 18 | 0 | 0 |
| Pachira aquatica | 0 | 0 | 0 | 0 |
| Paeonia lactiflora | 16535 | 14703 | 12871 | 9111 |
| Pandanus tectorius | 30 | 31 | 46 | 52 |
| Pandorea jasminoides | 1069 | 2061 | 1770 | 1238 |
| Pandorea pandorana | 1707 | 3606 | 3366 | 3493 |
| Papaver aculeatum | 2046 | 3201 | 2337 | 31 |
| Papaver nudicaule | 14792 | 11554 | 8645 | 6561 |
| Papaver orientale | 14116 | 13017 | 8794 | 9356 |
| Parmentiera aculeata | 0 | 0 | 0 | 0 |
| Parochetus communis | 224 | 293 | 205 | 882 |
| Passiflora amethystina | 88 | 257 | 73 | 25 |
| Passiflora coccinea | 0 | 0 | 0 | 0 |
| Passiflora foetida | 77 | 55 | 70 | 88 |
| Passiflora laurifolia | 0 | 0 | 0 | 0 |
| Passiflora ligularis | 1 | 13 | 0 | 13 |
| Passiflora mixta | 268 | 438 | 1 | 38 |
| Passiflora quadrangularis | 0 | 0 | 0 | 0 |
| Passiflora vitifolia | 0 | 0 | 0 | 0 |
| Pavonia hastata | 1481 | 3838 | 4135 | 9911 |
| Pelargonium | | | | |
| alchemilloides | 293 | 416 | 182 | 4 |
| Pelargonium | | | | |
| grossularioides | 1126 | 1818 | 1355 | 0 |
| Pennisetum alopecuroides | 7703 | 11494 | 14083 | 20239 |
| Pennisetum orientale | 1817 | 2036 | 2288 | 1079 |
| Penstemon gentianoides | 0 | 0 | 0 | 0 |
| Pentas lanceolata | 215 | 158 | 84 | 305 |
| Peperomia obtusifolia | 0 | 0 | 0 | 0 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|----------------------------|---------|--------|--------|--------|
| Pereskia aculeata | 282 | 463 | 500 | 373 |
| Perovskia atriplicifolia | 6015 | 7181 | 6328 | 2180 |
| Persea americana | 645 | 1210 | 602 | 1022 |
| Petrea volubilis | 5 | 4 | 6 | 14 |
| Petunia axillaris | 8043 | 10466 | 10462 | 16106 |
| Petunia integrifolia | 8579 | 9227 | 11054 | 8504 |
| Philadelphus mexicanus | 0 | 0 | 0 | 1 |
| Philadelphus pubescens | 4897 | 3118 | 767 | 0 |
| Philodendron | | | | |
| bipinnatifidum | 46 | 42 | 34 | 49 |
| Philodendron ornatum | 0 | 0 | 0 | 0 |
| Phlox drummondii | 11022 | 13487 | 12196 | 23059 |
| Phoenix reclinata | 75 | 63 | 43 | 12 |
| Photinia glabra | 422 | 660 | 1211 | 131 |
| Phyllanthus amarus | 0 | 0 | 0 | 0 |
| Phyllanthus emblica | 5 | 7 | 5 | 0 |
| Pimenta dioica | 0 | 0 | 0 | 0 |
| Pinellia ternata | 2827 | 4258 | 7586 | 21018 |
| Piper aduncum | 1 | 22 | 0 | 26 |
| Piscidia piscipula | 25 | 10 | 16 | 35 |
| Pistacia chinensis | 3530 | 5573 | 7277 | 8769 |
| Pithecellobium dulce | 0 | 1 | 22 | 6 |
| Pittosporum bicolor | 412 | 650 | 622 | 0 |
| Pittosporum eugenoides | 1137 | 1457 | 1210 | 2327 |
| Plantago rugelii | 279 | 868 | 885 | 3894 |
| Plectranthus amboinicus | 15 | 13 | 11 | 39 |
| Plectranthus ciliatus | 358 | 657 | 523 | 27 |
| Plectranthus | | | | |
| scutellarioides | 0 | 0 | 0 | 0 |
| Plectranthus verticillatus | 753 | 1289 | 872 | 702 |
| Plumbago zeylanica | 987 | 1084 | 1383 | 1331 |
| Plumeria obtusa | 0 | 0 | 0 | 0 |
| Plumeria rubra | 177 | 125 | 145 | 101 |
| Polygala lancifolia | 47 | 74 | 79 | 672 |
| Polygala senega | 203 | 112 | 83 | 1685 |
| Pomaderris lanigera | 1155 | 2197 | 2043 | 2198 |
| Populus acuminata | 0 | 0 | 0 | 0 |
| Portulacaria afra | 2323 | 2454 | 1862 | 226 |
| Pouteria caimito | 0 | 0 | 0 | 0 |
| Pratia repens | 65 | 129 | 26 | 0 |
| Prosopis chilensis | 254 | 326 | 274 | 325 |
| Prosopis juliflora | 358 | 492 | 817 | 43 |
| Prunus munsoniana | 4 | 272 | 1320 | 737 |
| Prunus pumila | 8 | 47 | 208 | 413 |
| Prunus salicina | 2812 | 3511 | 3985 | 4618 |
| Psoralea pinnata | 804 | 1682 | 1420 | 2053 |
| Pueraria montana | 1979 | 3185 | 4328 | 6183 |
| Pyracantha fortuneana | 3831 | 5059 | 5573 | 369 |
| Pyracantha koidzumii | 4697 | 7328 | 10126 | 14678 |
| Pyrus calleryana | 5777 | 7803 | 10607 | 15201 |
| Pyrus pyrifolia | 421 | 622 | 1031 | 419 |
| Quassia amara | 0 | 0 | 0 | 0 |
| Quercus acutissima | 758 | 1242 | 1693 | 1236 |
| Rauvolfia tetraphylla | 0 | 0 | 0 | 0 |
| Rauvolfia vomitoria | 0 | 0 | 0 | 0 |
| Rhamnus japonica | 124 | 123 | 275 | 36 |
| Rhaphiolepis indica | 2499 | 4002 | 5094 | 9111 |
| Rheum rhabarbarum | 18059 | 15050 | 12761 | 7148 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|-----------------------------|---------|--------|--------|--------|
| Rhipsalis baccifera | 8 | 10 | 6 | 28 |
| Rhodanthe chlorocephala | 3188 | 3376 | 1749 | 693 |
| Rhododendron japonicum | 1341 | 1954 | 2752 | 6938 |
| Romneya coulteri | 1529 | 2330 | 2027 | 224 |
| Romulea flava | 1446 | 1315 | 619 | 28 |
| Rosa banksiae | 4247 | 5218 | 6035 | 3316 |
| Rosa chinensis | 1992 | 2690 | 2744 | 279 |
| Rosenbergiodendron formosum | 0 | 0 | 0 | 0 |
| Rubus ellipticus | 1783 | 2645 | 2687 | 3737 |
| Rubus rosifolius | 2155 | 3956 | 4626 | 7399 |
| Rudbeckia triloba | 401 | 518 | 909 | 66 |
| Ruellia brevifolia | 156 | 251 | 277 | 367 |
| Ruellia tuberosa | 0 | 0 | 0 | 0 |
| Russelia equisetiformis | 240 | 175 | 129 | 408 |
| Russelia sarmentosa | 0 | 0 | 0 | 0 |
| Sageretia thea | 2808 | 3105 | 3298 | 3699 |
| Sagittaria graminea | 2568 | 4206 | 4632 | 6095 |
| Sagittaria montevidensis | 2185 | 5228 | 5921 | 8388 |
| Salix gracilistyla | 37 | 93 | 79 | 118 |
| Salix humboldtiana | 301 | 808 | 655 | 663 |
| Salix nigra | 1598 | 3253 | 3479 | 4818 |
| Salvia africana-lutea | 16 | 45 | 11 | 41 |
| Salvia leucantha | 490 | 872 | 602 | 969 |
| Salvia leucophylla | 379 | 454 | 441 | 0 |
| Salvia microphylla | 4026 | 5181 | 4641 | 1268 |
| Salvia plebeia | 3588 | 8140 | 11179 | 10230 |
| Salvia splendens | 4958 | 4736 | 4162 | 9846 |
| Sansevieria hyacinthoides | 153 | 214 | 197 | 11 |
| Santalum album | 0 | 0 | 0 | 0 |
| Sanvitalia procumbens | 1998 | 1932 | 519 | 758 |
| Sauropus androgynus | 136 | 281 | 252 | 159 |
| Scadoxus multiflorus | 72 | 64 | 30 | 5 |
| Scaevola taccada | 0 | 0 | 0 | 0 |
| Schefflera arboricola | 101 | 122 | 279 | 796 |
| Schisandra chinensis | 8 | 63 | 60 | 0 |
| Schotia brachypetala | 211 | 254 | 156 | 45 |
| Senecio radicans | 320 | 225 | 112 | 0 |
| Senecio tamoides | 314 | 493 | 515 | 0 |
| Senna artemisioides | 882 | 953 | 651 | 333 |
| Senna italica | 779 | 598 | 342 | 2 |
| Senna siamea | 0 | 2 | 39 | 1 |
| Sesbania grandiflora | 0 | 0 | 0 | 12 |
| Sesbania sesban | 131 | 126 | 109 | 0 |
| Sisyrinchium atlanticum | 1198 | 1773 | 2496 | 6637 |
| Sisyrinchium micranthum | 2285 | 2997 | 2302 | 2822 |
| Sisyrinchium mucronatum | 56 | 25 | 182 | 5585 |
| Solandra maxima | 65 | 53 | 42 | 78 |
| Solanum aviculare | 3881 | 4621 | 3969 | 4578 |
| Solanum betaceum | 5182 | 6713 | 3958 | 3806 |
| Solanum capsicoides | 599 | 1449 | 1380 | 1372 |
| Solanum chrysotrichum | 115 | 296 | 134 | 16 |
| Solanum lanceifolium | 0 | 0 | 0 | 0 |
| Solanum mammosum | 0 | 0 | 0 | 0 |
| Solanum quitoense | 0 | 0 | 0 | 0 |
| Solanum retroflexum | 694 | 977 | 307 | 1 |
| Solanum seaforthianum | 319 | 365 | 321 | 389 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|----------------------------|---------|--------|--------|--------|
| Solanum sessiliflorum | 0 | 0 | 0 | 0 |
| Solanum wendlandii | 23 | 26 | 23 | 49 |
| Solidago altissima | 3147 | 5933 | 6123 | 15496 |
| Solidago ptarmicoides | 478 | 1345 | 1347 | 5681 |
| Sophora tetraptera | 74 | 144 | 195 | 1385 |
| Sorbus alnifolia | 533 | 787 | 1151 | 1646 |
| Sparaxis grandiflora | 29 | 71 | 4 | 0 |
| Spathiphyllum cannifolium | 0 | 0 | 0 | 0 |
| Spathodea campanulata | 0 | 0 | 0 | 1 |
| Spathoglottis plicata | 0 | 0 | 0 | 0 |
| Spiraea prunifolia | 2430 | 4296 | 5856 | 12465 |
| Spondias dulcis | 0 | 0 | 0 | 3 |
| Spondias mombin | 0 | 0 | 0 | 0 |
| Stachytarpheta jamaicensis | 0 | 0 | 0 | 0 |
| Stachytarpheta mutabilis | 5 | 5 | 0 | 0 |
| Stapelia grandiflora | 14 | 4 | 0 | 0 |
| Stenocarpus sinuatus | 28 | 108 | 117 | 156 |
| Sterculia apetala | 0 | 0 | 0 | 0 |
| Stigmaphyllon ellipticum | 0 | 0 | 0 | 0 |
| Stipa tenuissima | 10092 | 11096 | 7167 | 7355 |
| Streptosolen jamesonii | 3 | 2 | 0 | 209 |
| Strophanthus gratus | 0 | 0 | 0 | 0 |
| Strophanthus preussii | 0 | 0 | 0 | 0 |
| Styrax japonicus | 196 | 227 | 522 | 1184 |
| Syagrus romanzoffiana | 333 | 515 | 547 | 1172 |
| Symplocos paniculata | 1410 | 1482 | 1741 | 735 |
| Syngonium angustatum | 0 | 0 | 0 | 0 |
| Syngonium podophyllum | 0 | 0 | 0 | 0 |
| Syringa reticulata | 2943 | 1310 | 2964 | 0 |
| Syzygium paniculatum | 247 | 500 | 397 | 80 |
| Tabebuia aurea | 0 | 0 | 5 | 0 |
| Tabernaemontana divaricata | 0 | 0 | 0 | 35 |
| Tacca leontopetaloides | 0 | 0 | 0 | 0 |
| Tagetes lucida | 0 | 0 | 17 | 36 |
| Tagetes tenuifolia | 3458 | 2954 | 333 | 278 |
| Tamarindus indica | 46 | 35 | 51 | 33 |
| Tamarix aphylla | 3964 | 4340 | 5017 | 4148 |
| Tamarix chinensis | 1915 | 1961 | 1839 | 1969 |
| Tanacetum coccineum | 10506 | 9597 | 6615 | 34 |
| Tecoma stans | 648 | 851 | 1033 | 1187 |
| Tephrosia candida | 18 | 22 | 25 | 161 |
| Tephrosia grandiflora | 373 | 746 | 732 | 919 |
| Tephrosia purpurea | 629 | 548 | 397 | 384 |
| Tephrosia vogelii | 0 | 1 | 0 | 0 |
| Terminalia catappa | 0 | 0 | 0 | 0 |
| Thalia geniculata | 0 | 0 | 0 | 0 |
| Theobroma cacao | 0 | 0 | 0 | 0 |
| Thunbergia alata | 802 | 1288 | 890 | 1342 |
| Thunbergia erecta | 0 | 0 | 0 | 10 |
| Thunbergia fragrans | 47 | 40 | 89 | 84 |
| Tigridia pavonia | 301 | 492 | 622 | 6 |
| Tillandsia stricta | 46 | 42 | 20 | 85 |
| Tillandsia usneoides | 1048 | 2127 | 2233 | 4416 |
| Tithonia rotundifolia | 4 | 5 | 27 | 0 |
| Toona ciliata | 409 | 664 | 710 | 1129 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|-----------------------------|---------|--------|--------|--------|
| Trachelospermum asiaticum | 421 | 659 | 1189 | 324 |
| Trachelospermum jasminoides | 8607 | 14689 | 16794 | 19063 |
| Tradescantia spathacea | 0 | 0 | 0 | 0 |
| Triadica sebifera | 1602 | 3002 | 4247 | 5889 |
| Trichocentrum carthagenense | 0 | 0 | 0 | 0 |
| Trichosanthes cucumerina | 0 | 0 | 0 | 0 |
| Trichosanthes kirilowii | 42 | 133 | 202 | 742 |
| Tripsacum dactyloides | 496 | 1530 | 1715 | 3298 |
| Tulbaghia violacea | 1122 | 1637 | 1081 | 278 |
| Ullucus tuberosus | 0 | 0 | 0 | 2 |
| Ulmus parvifolia | 5744 | 9461 | 11623 | 14695 |
| Ursinia anthemoides | 25 | 56 | 28 | 22 |
| Ursinia speciosa | 799 | 1660 | 1067 | 281 |
| Utricularia livida | 234 | 299 | 246 | 202 |
| Vallisneria americana | 7188 | 8459 | 8675 | 8454 |
| Vallisneria nana | 1595 | 2408 | 2239 | 2664 |
| Verbena stricta | 0 | 60 | 139 | 0 |
| Vernicia fordii | 807 | 1325 | 1643 | 1805 |
| Veronica americana | 9270 | 9532 | 7573 | 2879 |
| Viburnum dilatatum | 468 | 586 | 702 | 49 |
| Viburnum plicatum | 2808 | 2842 | 3780 | 11772 |

| species | current | RCP2.6 | RCP4.5 | RCP8.5 |
|--------------------------|---------|--------|--------|--------|
| Viburnum sieboldii | 613 | 907 | 1136 | 834 |
| Vigna caracalla | 154 | 189 | 135 | 373 |
| Viola hederacea | 568 | 916 | 1022 | 944 |
| Viola sororia | 5135 | 4613 | 4839 | 4741 |
| Vitex negundo | 1826 | 3637 | 4829 | 5597 |
| Vitis coignetiae | 4491 | 5960 | 6402 | 6946 |
| Washingtonia robusta | 2018 | 1748 | 1346 | 17 |
| Weigela floribunda | 358 | 161 | 81 | 0 |
| Westringia fruticosa | 1018 | 1828 | 2335 | 3195 |
| Wisteria floribunda | 4100 | 4672 | 6755 | 10119 |
| Xanthosoma sagittifolium | 11 | 7 | 3 | 35 |
| Ximenia americana | 111 | 96 | 74 | 2 |
| Zapoteca portoricensis | 2 | 0 | 0 | 0 |
| Zelkova serrata | 5209 | 6827 | 7879 | 18763 |
| Zingiber officinale | 0 | 0 | 4 | 2 |
| Zingiber zerumbet | 0 | 0 | 0 | 0 |
| Zinnia angustifolia | 3 | 0 | 0 | 12 |
| Zinnia elegans | 15472 | 19176 | 17972 | 15670 |
| Zinnia peruviana | 811 | 1046 | 996 | 212 |
| Ziziphus mauritiana | 30 | 49 | 111 | 0 |
| Ziziphus spina-christi | 81 | 54 | 64 | 0 |

SUPPORTING INFORMATION

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Appendix S2. Detailed model selection and downscaling procedure.

The three different scenarios (RCP2.6, RCP4.5 and RCP8.5) we used reflect different radiative forcing trajectories for the 21st century relative to pre-industrial conditions: The RCP2.6 scenario assumes that radiative forcing peaks at $\sim 3 \text{ W m}^{-2}$ before 2100 and then declines and is therefore referred to as mild scenario. In the intermediate scenario, RCP4.5, radiative forcing amounts to $\sim 4.5 \text{ W m}^{-2}$ at stabilization after 2100, while in the severe scenario, RCP8.5, radiative forcing continues to rise throughout the 21st century and reaches $> 8.5 \text{ W m}^{-2}$ in 2100 (Moss *et al.*, 2010). From all available models at the Cordex data portal (www.euro-cordex.net) future climate data (daily near surface temperature, monthly precipitation) were extracted. For each scenario, we then selected one model providing a relatively smooth time series of future climate parameters, namely: ICHEC-EC-EARTH_**rcp26**_r12i1p1_SMHI-RCA4, CNRM-CERFACS-CNRM-CM5_**rcp45**_r1i1p1_SMHI-RCA4, EUR-11_ICHEC-EC-EARTH_**rcp85**_r3i1p1_DMI-HIRHAM5, from now on referred to as RCP2.6, RCP4.5 and RCP8.5, respectively. Subsequent processing of these data included the following steps: (1) download of hindcast projections of the specific climate models, (2) deriving minimum, maximum and mean monthly temperatures from the daily values, (3) calculation of anomalies, i.e. temperature differences and precipitation quotients between future climate and their hindcast projections, (4) spatial interpolation of these anomalies to the 10' resolution surface using the natural neighbour method, and (5) addition or multiplication, respectively, of the interpolated temperature and precipitation anomalies to/with the 10' resolution current climate data from WorldClim. The resulting annual time series of future minimum, maximum and mean temperature and precipitation sums per month were averaged for the years 2050-2100 and the six bioclimatic variables selected for modelling were then recalculated from these average values. For further details on the downscaling methods see Dullinger *et al.* (2012).

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SUPPORTING INFORMATION

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Appendix S3. Selection and weighting of relevant CORINE land-cover classes for risk map assessment.

Using the CLC class descriptions and characteristics of the class contents from the CORINE land cover technical guide (EEA, 2000) we weighted CLC classes by their estimated amount of potential area for ornamental plant cultivation. As these estimates are necessarily imprecise but may have a considerable effect on the resulting risk maps, we used three separate weighting schemes that differed both in the average amount of planting area attributed to classes and in the relative weights given to each class (weighting schemes A, B, C). CLC classes with no potential ornamental area (e.g. pastures, forests and semi-natural areas) have been excluded.

Weighting scheme:

1. Artificial areas

A B C

1.1 Urban fabric

111 Continuous urban fabric

5 10 20

Most of the land is covered by structures and the transport network. Building, roads and artificially surfaced areas cover more than 80 % of the total surface. Non-linear areas of vegetation and bare soil are exceptional. Includes greenery (parks and grass areas) and small cemeteries <25ha.

112 Discontinuous urban fabric

10 15 20

Most of the land is covered by structures. Buildings, roads and artificially surfaced areas associated with vegetated areas and bare soil, which occupy discontinuous but significant surfaces. Includes parks, private gardens in suburbs, green spaces between blocks of flats, cemeteries <25ha, playgrounds.

1.2 Industrial, commercial and transport units

121 Industrial or commercial unit

0 1 5

Artificially surfaced areas (with concrete, asphalt, tarmacadam, or stabilised, e.g. beaten earth) without vegetation occupy most of the area, which also contains buildings and/or vegetation. Including stud farms, agricultural facilities (state farm centres).

| | | | |
|-----------------------------------|----------|----------|----------|
| 122 road and rail networks | 2 | 3 | 5 |
|-----------------------------------|----------|----------|----------|

Motorways and railways, including associated installations (stations, platforms, embankments). Minimum width for inclusion: 100 m.
Including linear greenery.

1.4 Artificial non-agricultural vegetated areas

| | | | |
|------------------------------|-----------|-----------|------------|
| 141 Green urban areas | 30 | 75 | 100 |
|------------------------------|-----------|-----------|------------|

Areas with vegetation within urban fabric, includes parks and cemeteries with vegetation, and mansions and their grounds.

| | | | |
|---|----------|----------|-----------|
| 142 Sport and leisure facilities | 2 | 5 | 20 |
|---|----------|----------|-----------|

Camping grounds, sports grounds, leisure parks, golf courses, racecourses, etc. Includes formal parks not surrounded by urban areas and cemeteries with vegetation situated outside of settlements, zoological and botanical gardens located outside of settlements, places of worship: e.g., convents, monasteries.

2. Agricultural areas

2.1 Arable land

| | | | |
|--------------------------------------|----------|----------|-----------|
| 211 Non-irrigated arable land | 1 | 2 | 10 |
|--------------------------------------|----------|----------|-----------|

Cereals, legumes, fodder crops, root crops and fallow land. Includes flowers and fruit trees (nurseries cultivation) and vegetables, whether open field, under plastic or glass (includes market gardening). Includes aromatic, medicinal and culinary plants, nurseries cultivation/gardens, and market gardening.

2.2 Permanent crops

| | | | |
|--|----------|----------|----------|
| 222 Fruit trees and berry plantations | 0 | 1 | 5 |
|--|----------|----------|----------|

Parcels planted with fruit trees or shrubs: single or mixed fruit species, fruit trees associated with permanently grassed surfaces, includes chestnut and walnut groves and plantations of Rosaceae.

2.4 Heterogeneous agricultural areas

| | | | |
|---|----------|----------|-----------|
| 242 Complex cultivation patterns | 1 | 2 | 10 |
|---|----------|----------|-----------|

Juxtaposition of small parcels of diverse annual crops, pastures and/or permanent crops, including hobby city gardens.

| | | | |
|--|----------|----------|----------|
| 243 Land occupied by agriculture, with significant natural vegetation | 1 | 2 | 5 |
|--|----------|----------|----------|

Areas principally occupied by agriculture, interspersed with significant natural areas, includes sporadically occurring houses of rural settlements or farm buildings and their gardens.

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Appendix S4. Information on model performance.

The table lists for all species and each modelling technique the mean TSS over all replicates and the percentage of replicates with TSS < 0.5.

| species \ model | GLM | | GAM | | RF | | GBM | |
|----------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Abelmoschus moschatus | 0.823 | 0 | 0.823 | 0 | 0.856 | 0 | 0.847 | 0 |
| Abrus precatorius | 0.823 | 0 | 0.829 | 0 | 0.876 | 0 | 0.863 | 0 |
| Acacia pravissima | 0.968 | 0 | 0.934 | 0 | 0.943 | 0 | 0.941 | 0 |
| Acalypha hispida | 0.736 | 0 | 0.708 | 0 | 0.804 | 0 | 0.802 | 0 |
| Acalypha wilkesiana | 0.759 | 0 | 0.689 | 0 | 0.809 | 0 | 0.794 | 0 |
| Acanthocereus tetragonus | 0.895 | 0 | 0.888 | 0 | 0.916 | 0 | 0.917 | 0 |
| Acer buergerianum | 0.827 | 0 | 0.783 | 0 | 0.891 | 0 | 0.864 | 3.3 |
| Acer japonicum | 0.871 | 0 | 0.937 | 0 | 0.922 | 0 | 0.902 | 0 |
| Acer palmatum | 0.884 | 0 | 0.881 | 0 | 0.921 | 0 | 0.909 | 0 |
| Acer spicatum | 0.925 | 0 | 0.938 | 0 | 0.962 | 0 | 0.959 | 0 |
| Actinidia arguta | 0.904 | 0 | 0.858 | 0 | 0.945 | 0 | 0.944 | 0 |
| Actinidia polygama | 0.96 | 0 | 0.907 | 0 | 0.947 | 0 | 0.948 | 0 |
| Adansonia digitata | 0.808 | 0 | 0.761 | 0 | 0.825 | 0 | 0.824 | 0 |
| Adenanthura pavonina | 0.813 | 0 | 0.758 | 0 | 0.876 | 0 | 0.868 | 0 |
| Adenium obesum | 0.841 | 0 | 0.805 | 0 | 0.844 | 0 | 0.846 | 0 |
| Aesculus glabra | 0.911 | 0 | 0.888 | 0 | 0.931 | 0 | 0.922 | 0 |
| Agastache rugosa | 0.864 | 0 | 0.803 | 0 | 0.862 | 0 | 0.848 | 0 |
| Ageratina ligustrina | 0.907 | 0 | 0.91 | 0 | 0.923 | 0 | 0.917 | 0 |
| Ageratum conyzoides | 0.759 | 0 | 0.761 | 0 | 0.874 | 0 | 0.852 | 0 |
| Albizia chinensis | 0.878 | 0 | 0.823 | 0 | 0.87 | 0 | 0.87 | 0 |
| Albizia lebeck | 0.723 | 0 | 0.75 | 0 | 0.793 | 0 | 0.787 | 0 |
| Albizia saman | 0.807 | 0 | 0.796 | 0 | 0.852 | 0 | 0.85 | 0 |
| Albica bracteata | 0.749 | 0 | 0.898 | 0 | 0.906 | 0 | 0.922 | 0 |
| Albica canadensis | 0.937 | 0 | 0.873 | 33.3 | 0.972 | 0 | 0.956 | 0 |
| Aleurites moluccana | 0.849 | 0 | 0.81 | 0 | 0.848 | 0 | 0.848 | 0 |
| Allamanda cathartica | 0.823 | 0 | 0.828 | 0 | 0.873 | 0 | 0.871 | 0 |
| Allium tuberosum | 0.765 | 0 | 0.707 | 0 | 0.807 | 0 | 0.795 | 0 |
| Allocastraria littoralis | 0.967 | 0 | 0.979 | 0 | 0.98 | 0 | 0.973 | 0 |
| Allocastraria verticillata | 0.964 | 0 | 0.969 | 0 | 0.974 | 0 | 0.97 | 0 |
| Alnus nepalensis | 0.94 | 0 | 0.929 | 0 | 0.952 | 0 | 0.946 | 0 |
| Alocasia macrorrhizos | 0.835 | 0 | 0.837 | 0 | 0.861 | 0 | 0.847 | 0 |
| Alstonia scholaris | 0.882 | 0 | 0.785 | 0 | 0.842 | 0 | 0.844 | 0 |
| Alternanthera ficoidea | 0.754 | 0 | 0.815 | 0 | 0.837 | 0 | 0.83 | 0 |
| Ammobium alatum | 0.845 | 0 | 0.831 | 0 | 0.944 | 0 | 0.94 | 0 |
| Ampelopsis glandulosa | 0.9 | 0 | 0.86 | 0 | 0.944 | 0 | 0.937 | 0 |
| Anacardium occidentale | 0.811 | 0 | 0.818 | 0 | 0.889 | 0 | 0.885 | 0 |
| Ananas comosus | 0.717 | 0 | 0.73 | 0 | 0.802 | 0 | 0.781 | 0 |
| Angelonia angustifolia | 0.839 | 0 | 0.774 | 0 | 0.882 | 0 | 0.885 | 0 |
| Anigozanthos flavidus | 0.807 | 0 | 0.907 | 0 | 0.955 | 0 | 0.942 | 0 |
| Annona cherimola | 0.786 | 0 | 0.834 | 0 | 0.866 | 0 | 0.852 | 0 |
| Annona glabra | 0.845 | 0 | 0.872 | 0 | 0.885 | 0 | 0.878 | 0 |
| Annona muricata | 0.823 | 0 | 0.827 | 0 | 0.861 | 0 | 0.855 | 0 |

| species | model | GLM | | GAM | | RF | | GBM | |
|--------------------------------|-------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Annona reticulata | | 0.812 | 0 | 0.826 | 0 | 0.879 | 0 | 0.873 | 0 |
| Annona squamosa | | 0.807 | 0 | 0.799 | 0 | 0.822 | 0 | 0.813 | 0 |
| Anoda cristata | | 0.677 | 0 | 0.75 | 0 | 0.845 | 0 | 0.794 | 0 |
| Anthurium pentaphyllum | | 0.887 | 0 | 0.873 | 0 | 0.914 | 0 | 0.915 | 0 |
| Anthurium schlechtendalii | | 0.885 | 0 | 0.924 | 0 | 0.924 | 0 | 0.923 | 0 |
| Antigonon leptopus | | 0.733 | 0 | 0.75 | 0 | 0.816 | 0 | 0.799 | 0 |
| Apodytes dimidiata | | 0.858 | 0 | 0.883 | 0 | 0.925 | 0 | 0.907 | 0 |
| Archontophoenix cunninghamiana | | 0.94 | 0 | 0.912 | 0 | 0.967 | 0 | 0.963 | 0 |
| Arctotis fastuosa | | 0.884 | 0 | 0.88 | 0 | 0.944 | 0 | 0.944 | 0 |
| Ardisia crenata | | 0.882 | 0 | 0.873 | 0 | 0.897 | 0 | 0.894 | 0 |
| Ardisia japonica | | 0.965 | 0 | 0.953 | 0 | 0.971 | 0 | 0.967 | 0 |
| Aristolochia grandiflora | | 0.84 | 0 | 0.819 | 0 | 0.856 | 0 | 0.854 | 0 |
| Artemisia ludoviciana | | 0.778 | 0 | 0.81 | 0 | 0.905 | 0 | 0.842 | 0 |
| Artocarpus altilis | | 0.849 | 0 | 0.829 | 0 | 0.872 | 0 | 0.868 | 0 |
| Artocarpus heterophyllus | | 0.767 | 0 | 0.784 | 0 | 0.854 | 0 | 0.836 | 0 |
| Arum palaestinum | | 0.874 | 0 | 0.846 | 0 | 0.945 | 0 | 0.93 | 0 |
| Arundina graminifolia | | 0.856 | 0 | 0.897 | 0 | 0.892 | 0 | 0.887 | 0 |
| Asimina triloba | | 0.949 | 0 | 0.956 | 0 | 0.963 | 0 | 0.954 | 0 |
| Asparagus africanus | | 0.816 | 0 | 0.817 | 0 | 0.862 | 0 | 0.848 | 0 |
| Asparagus declinatus | | 0.937 | 0 | 0.89 | 0 | 0.937 | 0 | 0.92 | 0 |
| Asparagus falcatus | | 0.826 | 0 | 0.846 | 0 | 0.872 | 0 | 0.854 | 0 |
| Asparagus retrofractus | | 0.947 | 0 | 0.96 | 0 | 0.959 | 0 | 0.951 | 0 |
| Asparagus scandens | | 0.977 | 0 | 0.9 | 0 | 0.964 | 0 | 0.958 | 0 |
| Asparagus virgatus | | 0.917 | 0 | 0.913 | 33.3 | 0.954 | 0 | 0.936 | 0 |
| Atriplex canescens | | 0.78 | 0 | 0.805 | 0 | 0.917 | 0 | 0.872 | 0 |
| Atriplex nummularia | | 0.908 | 0 | 0.91 | 0 | 0.958 | 0 | 0.952 | 0 |
| Averrhoa carambola | | 0.831 | 0 | 0.754 | 0 | 0.791 | 0 | 0.794 | 0 |
| Banisteriopsis caapi | | 0.801 | 0 | 0.708 | 0 | 0.912 | 0 | 0.906 | 0 |
| Banksia ericifolia | | 0.961 | 0 | 0.935 | 0 | 0.987 | 0 | 0.975 | 0 |
| Barleria cristata | | 0.673 | 0 | 0.618 | 0 | 0.769 | 6.7 | 0.774 | 3.3 |
| Barringtonia asiatica | | 0.914 | 0 | 0.824 | 0 | 0.918 | 0 | 0.909 | 0 |
| Bartlettina sordida | | 0.791 | 0 | 0.753 | 0 | 0.857 | 0 | 0.842 | 0 |
| Basella alba | | 0.72 | 0 | 0.766 | 0 | 0.809 | 0 | 0.805 | 0 |
| Bauhinia galpinii | | 0.894 | 0 | 0.865 | 0 | 0.919 | 0 | 0.907 | 0 |
| Bauhinia purpurea | | 0.741 | 0 | 0.678 | 0 | 0.815 | 0 | 0.81 | 0 |
| Bauhinia variegata | | 0.748 | 0 | 0.763 | 0 | 0.77 | 0 | 0.773 | 0 |
| Begonia cucullata | | 0.782 | 0 | 0.733 | 0 | 0.858 | 0 | 0.848 | 0 |
| Begonia heracleifolia | | 0.883 | 0 | 0.844 | 0 | 0.927 | 0 | 0.911 | 0 |
| Begonia humilis | | 0.847 | 0 | 0.873 | 0 | 0.902 | 0 | 0.895 | 0 |
| Begonia nelumbiifolia | | 0.844 | 0 | 0.786 | 0 | 0.879 | 0 | 0.875 | 0 |
| Berberis glaucocarpa | | 0.92 | 0 | 0.801 | 0 | 0.961 | 0 | 0.942 | 0 |
| Berberis repens | | 0.916 | 0 | 0.922 | 0 | 0.947 | 0 | 0.943 | 0 |
| Bertholletia excelsa | | 0.942 | 0 | 0.741 | 33.3 | 0.925 | 0 | 0.923 | 0 |
| Betula platyphylla | | NaN | 100 | 0.851 | 0 | 0.881 | 0 | 0.873 | 0 |
| Bignonia capreolata | | 0.939 | 0 | 0.917 | 0 | 0.95 | 0 | 0.945 | 0 |
| Billardiera heterophylla | | 0.866 | 0 | 0.886 | 0 | 0.89 | 0 | 0.883 | 0 |
| Bixa orellana | | 0.807 | 0 | 0.801 | 0 | 0.881 | 0 | 0.877 | 0 |
| Bocconia frutescens | | 0.818 | 0 | 0.816 | 0 | 0.913 | 0 | 0.905 | 0 |
| Boltonia asteroides | | 0.849 | 0 | 0.862 | 0 | 0.929 | 0 | 0.914 | 0 |
| Bomarea multiflora | | 0.957 | 0 | 0.936 | 0 | 0.966 | 0 | 0.97 | 0 |
| Bombax ceiba | | 0.854 | 0 | 0.889 | 0 | 0.909 | 0 | 0.892 | 0 |
| Bouteloua curtipendula | | 0.753 | 0 | 0.792 | 0 | 0.904 | 0 | 0.847 | 0 |
| Brachychiton acerifolius | | 0.95 | 0 | 0.946 | 0 | 0.932 | 0 | 0.929 | 0 |
| Brachychiton discolor | | 0.771 | 0 | 0.871 | 0 | 0.872 | 0 | 0.871 | 0 |
| Brachychiton populneus | | 0.965 | 0 | 0.963 | 0 | 0.975 | 0 | 0.97 | 0 |
| Brexia madagascariensis | | 0.916 | 0 | 0.916 | 0 | 0.942 | 0 | 0.944 | 0 |
| Breynia disticha | | 0.77 | 0 | 0.791 | 0 | 0.86 | 0 | 0.857 | 0 |
| Bromus briziformis | | 0.954 | 0 | 0.936 | 0 | 0.925 | 0 | 0.917 | 0 |
| Bromus danthoniae | | 0.845 | 0 | 0.91 | 0 | 0.885 | 0 | 0.876 | 0 |
| Browallia americana | | 0.83 | 0 | 0.871 | 0 | 0.909 | 0 | 0.903 | 0 |
| Brownea coccinea | | 0.933 | 0 | 0.898 | 0 | 0.933 | 0 | 0.915 | 0 |
| Brownea grandiceps | | 0.88 | 0 | 0.821 | 0 | 0.944 | 0 | 0.936 | 0 |
| Brugmansia sanguinea | | 0.884 | 0 | 0.846 | 0 | 0.963 | 0 | 0.95 | 0 |
| Brunfelsia uniflora | | 0.721 | 0 | 0.791 | 0 | 0.847 | 0 | 0.822 | 0 |

| species | model | GLM | | GAM | | RF | | GBM | |
|----------------------------|-------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Bucida buceras | | 0.875 | 0 | 0.885 | 0 | 0.928 | 0 | 0.922 | 0 |
| Buddleja asiatica | | 0.823 | 0 | 0.817 | 0 | 0.842 | 0 | 0.836 | 0 |
| Buddleja indica | | 0.968 | 0 | 0.945 | 0 | 0.972 | 0 | 0.96 | 0 |
| Buddleja saligna | | 0.95 | 0 | 0.941 | 0 | 0.942 | 0 | 0.927 | 0 |
| Buddleja salviifolia | | 0.937 | 0 | 0.939 | 0 | 0.952 | 0 | 0.937 | 0 |
| Buddleja stachyoides | | 0.744 | 0 | 0.821 | 0 | 0.936 | 0 | 0.904 | 0 |
| Bulbine semibarbata | | 0.921 | 0 | 0.924 | 0 | 0.961 | 0 | 0.953 | 0 |
| Bursera simaruba | | 0.813 | 0 | 0.826 | 0 | 0.863 | 0 | 0.846 | 0 |
| Caesalpinia coriaria | | 0.84 | 0 | 0.869 | 0 | 0.871 | 0 | 0.87 | 0 |
| Caesalpinia pulcherrima | | 0.691 | 0 | 0.723 | 0 | 0.816 | 0 | 0.801 | 0 |
| Caladium bicolor | | 0.776 | 0 | 0.822 | 0 | 0.872 | 0 | 0.867 | 0 |
| Calceolaria chelidonioides | | 0.866 | 0 | 0.814 | 0 | 0.902 | 0 | 0.909 | 0 |
| Calliandra haematocephala | | 0.798 | 0 | 0.78 | 0 | 0.82 | 0 | 0.798 | 0 |
| Calliandra houstoniana | | 0.838 | 0 | 0.889 | 0 | 0.91 | 0 | 0.891 | 0 |
| Calliandra surinamensis | | 0.842 | 0 | 0.808 | 0 | 0.9 | 0 | 0.903 | 0 |
| Callicarpa dichotoma | | 0.902 | 0 | 0.828 | 0 | 0.955 | 0 | 0.93 | 0 |
| Callicarpa japonica | | 0.959 | 0 | 0.939 | 0 | 0.96 | 0 | 0.956 | 0 |
| Callisia repens | | 0.771 | 0 | 0.783 | 0 | 0.837 | 0 | 0.825 | 0 |
| Callistemon speciosus | | 0.756 | 0 | 0.816 | 0 | 0.861 | 0 | 0.831 | 0 |
| Callistemon viminalis | | 0.818 | 0 | 0.856 | 0 | 0.9 | 0 | 0.886 | 0 |
| Calophyllum inophyllum | | 0.93 | 0 | 0.932 | 0 | 0.912 | 0 | 0.91 | 0 |
| Calotropis procera | | 0.699 | 0 | 0.746 | 0 | 0.807 | 0 | 0.772 | 0 |
| Calystegia hederacea | | 0.935 | 0 | 0.918 | 0 | 0.974 | 0 | 0.969 | 0 |
| Camellia japonica | | 0.895 | 0 | 0.912 | 0 | 0.913 | 0 | 0.912 | 0 |
| Camellia sinensis | | 0.916 | 0 | 0.856 | 0 | 0.922 | 0 | 0.914 | 0 |
| Campanula punctata | | 0.901 | 0 | 0.898 | 0 | 0.946 | 0 | 0.94 | 0 |
| Cananga odorata | | 0.875 | 0 | 0.863 | 0 | 0.864 | 0 | 0.857 | 0 |
| Canavalia cathartica | | 0.805 | 0 | 0.76 | 0 | 0.837 | 0 | 0.823 | 0 |
| Canavalia ensiformis | | 0.762 | 0 | 0.713 | 0 | 0.75 | 0 | 0.75 | 0 |
| Canna glauca | | 0.706 | 0 | 0.697 | 0 | 0.813 | 0 | 0.802 | 0 |
| Cardiocrinum cordatum | | 0.977 | 0 | 0.96 | 0 | 0.996 | 0 | 0.991 | 0 |
| Carica papaya | | 0.758 | 0 | 0.768 | 0 | 0.832 | 0 | 0.821 | 0 |
| Carissa macrocarpa | | 0.649 | 0 | 0.596 | 33.3 | 0.813 | 0 | 0.773 | 0 |
| Carpinus caroliniana | | 0.843 | 0 | 0.87 | 0 | 0.932 | 0 | 0.916 | 0 |
| Carya illinoensis | | 0.849 | 0 | 0.851 | 0 | 0.87 | 0 | 0.859 | 0 |
| Cascabela thevetia | | 0.8 | 0 | 0.727 | 0 | 0.773 | 3.3 | 0.73 | 3.3 |
| Cassia fistula | | 0.725 | 0 | 0.704 | 0 | 0.801 | 0 | 0.792 | 0 |
| Cassia grandis | | 0.801 | 0 | 0.807 | 0 | 0.878 | 0 | 0.876 | 0 |
| Castanea mollissima | | 0.885 | 0 | 0.835 | 0 | 0.923 | 0 | 0.906 | 0 |
| Castanospermum australe | | 0.954 | 0 | 0.903 | 0 | 0.938 | 0 | 0.929 | 0 |
| Casuarina cunninghamiana | | 0.853 | 0 | 0.854 | 0 | 0.922 | 0 | 0.897 | 0 |
| Casuarina glauca | | 0.967 | 0 | 0.934 | 0 | 0.976 | 0 | 0.975 | 0 |
| Cedrela odorata | | 0.8 | 0 | 0.8 | 0 | 0.881 | 0 | 0.869 | 0 |
| Ceiba pentandra | | 0.782 | 0 | 0.792 | 0 | 0.857 | 0 | 0.854 | 0 |
| Celastrus scandens | | 0.912 | 0 | 0.926 | 0 | 0.968 | 0 | 0.957 | 0 |
| Centrosema virginianum | | 0.735 | 0 | 0.769 | 0 | 0.859 | 0 | 0.84 | 0 |
| Ceratopetalum gummiiferum | | 0.985 | 0 | 0.985 | 0 | 0.994 | 0 | 0.986 | 0 |
| Ceratotheca triloba | | NaN | 100 | 0.933 | 0 | 0.976 | 0 | 0.957 | 0 |
| Cercestis mirabilis | | 0.931 | 0 | 0.932 | 0 | 0.951 | 0 | 0.945 | 0 |
| Cercidiphyllum japonicum | | 0.859 | 0 | 0.85 | 0 | 0.912 | 0 | 0.908 | 0 |
| Cestrum aurantiacum | | 0.81 | 0 | 0.763 | 0 | 0.875 | 0 | 0.865 | 0 |
| Cestrum fasciculatum | | 0.742 | 33.3 | 0.825 | 0 | 0.905 | 0 | 0.887 | 0 |
| Cestrum nocturnum | | 0.819 | 0 | 0.821 | 0 | 0.863 | 0 | 0.844 | 0 |
| Chamaedorea elegans | | 0.879 | 0 | 0.781 | 0 | 0.92 | 0 | 0.907 | 0 |
| Chamelaucium uncinatum | | 0.808 | 0 | 0.917 | 0 | 0.953 | 0 | 0.931 | 0 |
| Cheilocostus speciosus | | 0.837 | 0 | 0.845 | 0 | 0.877 | 0 | 0.879 | 0 |
| Chenopodium quinoa | | 0.837 | 0 | 0.77 | 0 | 0.875 | 0 | 0.858 | 0 |
| Chrysobalanus icaco | | 0.844 | 0 | 0.84 | 0 | 0.9 | 0 | 0.891 | 0 |
| Chrysophyllum cainito | | 0.845 | 0 | 0.826 | 0 | 0.852 | 0 | 0.84 | 0 |
| Chrysophyllum oliviforme | | 0.859 | 0 | 0.932 | 0 | 0.915 | 0 | 0.892 | 0 |
| Chrysothemis pulchella | | 0.91 | 0 | 0.886 | 0 | 0.924 | 0 | 0.924 | 0 |
| Cinnamomum camphora | | 0.871 | 0 | 0.862 | 0 | 0.905 | 0 | 0.902 | 0 |
| Cissus alata | | 0.878 | 0 | 0.729 | 0 | 0.91 | 0 | 0.914 | 0 |
| Cissus antarctica | | 0.995 | 0 | 0.985 | 0 | 0.994 | 0 | 0.991 | 0 |
| Cissus quadrangularis | | 0.862 | 0 | 0.859 | 0 | 0.863 | 0 | 0.863 | 0 |

| species | GLM | | GAM | | RF | | GBM | |
|---------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Cissus rotundifolia | 0.884 | 0 | 0.868 | 0 | 0.886 | 0 | 0.884 | 0 |
| Cissus verticillata | 0.785 | 0 | 0.773 | 0 | 0.89 | 0 | 0.868 | 0 |
| Citharexylum spinosum | 0.863 | 0 | 0.817 | 0 | 0.898 | 0 | 0.89 | 0 |
| Citrus aurantiifolia | 0.793 | 0 | 0.772 | 0 | 0.817 | 0 | 0.812 | 0 |
| Citrus maxima | 0.717 | 0 | 0.727 | 0 | 0.811 | 0 | 0.797 | 0 |
| Clarkia amoena | 0.856 | 0 | 0.898 | 0 | 0.942 | 0 | 0.927 | 0 |
| Clarkia pulchella | 0.886 | 0 | 0.893 | 0 | 0.925 | 0 | 0.915 | 0 |
| Clematis paniculata | 0.966 | 0 | 0.945 | 0 | 0.973 | 0 | 0.964 | 0 |
| Clematis tangutica | 0.84 | 0 | 0.848 | 0 | 0.925 | 0 | 0.914 | 0 |
| Clematis terniflora | 0.832 | 0 | 0.898 | 0 | 0.928 | 0 | 0.915 | 0 |
| Cleome gynandra | 0.704 | 0 | 0.729 | 0 | 0.797 | 0 | 0.772 | 0 |
| Clerodendrum bungei | 0.918 | 0 | 0.837 | 0 | 0.833 | 0 | 0.813 | 0 |
| Clerodendrum chinense | 0.809 | 0 | 0.79 | 0 | 0.847 | 0 | 0.835 | 0 |
| Clerodendrum splendens | 0.897 | 0 | 0.887 | 0 | 0.917 | 0 | 0.911 | 0 |
| Clerodendrum trichotomum | 0.921 | 0 | 0.905 | 0 | 0.934 | 0 | 0.926 | 0 |
| Cleyera japonica | 0.952 | 0 | 0.952 | 0 | 0.957 | 0 | 0.948 | 0 |
| Clidemia hirta | 0.857 | 0 | 0.853 | 0 | 0.925 | 0 | 0.919 | 0 |
| Clitoria ternatea | 0.74 | 0 | 0.739 | 0 | 0.832 | 0 | 0.823 | 0 |
| Clusia rosea | 0.837 | 0 | 0.844 | 0 | 0.881 | 0 | 0.881 | 0 |
| Cobaea scandens | 0.875 | 0 | 0.792 | 0 | 0.917 | 0 | 0.912 | 0 |
| Coccinia grandis | 0.718 | 0 | 0.719 | 0 | 0.737 | 0 | 0.739 | 0 |
| Coccoloba uvifera | 0.835 | 0 | 0.879 | 0 | 0.857 | 0 | 0.856 | 0 |
| Cochlospermum vitifolium | 0.82 | 0 | 0.819 | 0 | 0.862 | 0 | 0.856 | 0 |
| Cocos nucifera | 0.817 | 0 | 0.806 | 0 | 0.814 | 0 | 0.819 | 0 |
| Codiaeum variegatum | 0.848 | 0 | 0.831 | 0 | 0.891 | 0 | 0.881 | 0 |
| Coffea liberica | 0.86 | 0 | 0.831 | 0 | 0.91 | 0 | 0.903 | 0 |
| Cola acuminata | 0.917 | 0 | 0.853 | 0 | 0.867 | 0 | 0.863 | 0 |
| Combretum grandiflorum | 0.952 | 0 | 0.915 | 0 | 0.909 | 0 | 0.9 | 0 |
| Combretum indicum | 0.757 | 0 | 0.747 | 0 | 0.785 | 0 | 0.782 | 0 |
| Commelina benghalensis | 0.669 | 0 | 0.712 | 0 | 0.806 | 0 | 0.768 | 0 |
| Commelina coelestis | 0.887 | 0 | 0.846 | 0 | 0.905 | 0 | 0.9 | 0 |
| Conicosia pugioniformis | 0.988 | 0 | 0.969 | 0 | 0.968 | 0 | 0.951 | 0 |
| Coprosma robusta | 0.981 | 0 | 0.941 | 0 | 0.993 | 0 | 0.987 | 0 |
| Corchorus olitorius | 0.741 | 0 | 0.746 | 0 | 0.798 | 0 | 0.78 | 0 |
| Cordyline fruticosa | 0.807 | 0 | 0.732 | 0 | 0.839 | 0 | 0.844 | 0 |
| Cortaderia jubata | 0.86 | 0 | 0.823 | 0 | 0.918 | 0 | 0.907 | 0 |
| Corylus heterophylla | 0.766 | 0 | 0.783 | 0 | 0.903 | 0 | 0.892 | 0 |
| Corymbia citriodora | 0.858 | 0 | 0.863 | 0 | 0.895 | 0 | 0.875 | 0 |
| Cosmos sulphureus | 0.683 | 0 | 0.701 | 0 | 0.791 | 0 | 0.771 | 0 |
| Cotoneaster glaucophyllus | 0.942 | 0 | 0.925 | 0 | 0.968 | 0 | 0.956 | 0 |
| Couroupita guianensis | 0.904 | 0 | 0.859 | 0 | 0.913 | 0 | 0.926 | 0 |
| Crassula ericoides | 0.968 | 0 | 0.886 | 0 | 0.961 | 0 | 0.95 | 0 |
| Crataegus pubescens | 0.935 | 0 | 0.919 | 0 | 0.944 | 0 | 0.937 | 0 |
| Crescentia cujete | 0.803 | 0 | 0.81 | 0 | 0.853 | 0 | 0.849 | 0 |
| Crinum asiaticum | 0.872 | 0 | 0.853 | 0 | 0.869 | 0 | 0.873 | 0 |
| Crinum zeylanicum | 0.864 | 0 | 0.837 | 0 | 0.853 | 0 | 0.856 | 0 |
| Crotalaria capensis | 0.944 | 0 | 0.921 | 0 | 0.952 | 0 | 0.942 | 0 |
| Crotalaria juncea | 0.738 | 0 | 0.754 | 0 | 0.803 | 0 | 0.786 | 0 |
| Crotalaria micans | 0.778 | 0 | 0.791 | 0 | 0.873 | 0 | 0.859 | 0 |
| Crotalaria retusa | 0.78 | 0 | 0.793 | 0 | 0.868 | 0 | 0.864 | 0 |
| Cucumis anguria | 0.775 | 0 | 0.793 | 0 | 0.823 | 0 | 0.811 | 0 |
| Cucumis dipsaceus | 0.706 | 0 | 0.664 | 0 | 0.82 | 0 | 0.833 | 0 |
| Cucumis metuliferus | 0.703 | 0 | 0.781 | 0 | 0.784 | 0 | 0.779 | 0 |
| Cucurbita argyrosperma | 0.795 | 0 | 0.811 | 0 | 0.839 | 0 | 0.829 | 0 |
| Cucurbita ficifolia | 0.778 | 0 | 0.777 | 0 | 0.802 | 0 | 0.793 | 0 |
| Cucurbita moschata | 0.648 | 0 | 0.677 | 0 | 0.766 | 0 | 0.733 | 0 |
| Cuphea hyssopifolia | 0.79 | 0 | 0.811 | 0 | 0.83 | 0 | 0.815 | 0 |
| Curcuma longa | 0.881 | 0 | 0.792 | 0 | 0.844 | 0 | 0.858 | 0 |
| Cymbopogon nardus | 0.892 | 0 | 0.883 | 0 | 0.906 | 0 | 0.9 | 0 |
| Cynoglossum amabile | 0.799 | 0 | 0.838 | 0 | 0.821 | 0 | 0.806 | 0 |
| Cynoglossum zeylanicum | 0.871 | 0 | 0.852 | 0 | 0.878 | 0 | 0.872 | 0 |
| Cyperus albstriatus | 0.921 | 0 | 0.913 | 0 | 0.946 | 0 | 0.933 | 0 |
| Cytisus proliferus | 0.901 | 0 | 0.909 | 0 | 0.942 | 0 | 0.935 | 0 |
| Dahlia imperialis | 0.912 | 0 | 0.917 | 0 | 0.923 | 0 | 0.92 | 0 |
| Dahlia pinnata | 0.814 | 0 | 0.858 | 0 | 0.886 | 0 | 0.874 | 0 |

| species | model | GLM | | GAM | | RF | | GBM | |
|-----------------------------|-------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Datura ceratocaula | | 0.925 | 0 | 0.88 | 0 | 0.896 | 0 | 0.894 | 0 |
| Debregeasia longifolia | | 0.804 | 0 | 0.812 | 0 | 0.902 | 0 | 0.884 | 0 |
| Delonix regia | | 0.752 | 0 | 0.72 | 0 | 0.787 | 0 | 0.777 | 0 |
| Desmodium elegans | | 0.843 | 0 | 0.878 | 0 | 0.921 | 0 | 0.904 | 0 |
| Deutzia crenata | | 0.94 | 0 | 0.96 | 0 | 0.962 | 0 | 0.956 | 0 |
| Deutzia gracilis | | 0.913 | 0 | 0.859 | 0 | 0.942 | 0 | 0.928 | 0 |
| Dianella ensifolia | | 0.881 | 0 | 0.865 | 0 | 0.881 | 0 | 0.871 | 0 |
| Dieffenbachia seguine | | 0.827 | 0 | 0.809 | 0 | 0.919 | 0 | 0.91 | 0 |
| Dimorphotheca cuneata | | 0.957 | 0 | 0.966 | 0 | 0.969 | 0 | 0.955 | 0 |
| Dimorphotheca pluvialis | | 0.869 | 0 | 0.791 | 0 | 0.907 | 0 | 0.895 | 0 |
| Dimorphotheca sinuata | | 0.855 | 0 | 0.882 | 0 | 0.904 | 0 | 0.891 | 0 |
| Dioscorea bulbifera | | 0.776 | 0 | 0.819 | 0 | 0.867 | 0 | 0.853 | 0 |
| Dioscorea mexicana | | 0.842 | 0 | 0.806 | 0 | 0.883 | 0 | 0.879 | 0 |
| Dioscorea polystachya | | 0.926 | 0 | 0.749 | 0 | 0.91 | 0 | 0.918 | 0 |
| Diospyros virginiana | | 0.915 | 0 | 0.928 | 0 | 0.945 | 0 | 0.931 | 0 |
| Diplocyclos palmatus | | 0.833 | 0 | 0.799 | 0 | 0.865 | 0 | 0.849 | 0 |
| Dodonaea viscosa | | 0.852 | 0 | 0.874 | 0 | 0.922 | 0 | 0.914 | 0 |
| Dolichandra unguis-cati | | 0.729 | 0 | 0.756 | 0 | 0.857 | 0 | 0.838 | 0 |
| Dombeya burgessiae | | 0.906 | 0 | 0.849 | 0 | 0.907 | 0 | 0.892 | 0 |
| Dombeya tiliacea | | 0.829 | 0 | 0.795 | 0 | 0.967 | 0 | 0.95 | 0 |
| Dorotheanthus bellidiformis | | 0.939 | 0 | 0.946 | 0 | 0.978 | 0 | 0.969 | 0 |
| Dorstenia contrajerva | | 0.819 | 0 | 0.853 | 0 | 0.891 | 0 | 0.882 | 0 |
| Dracaena fragrans | | 0.775 | 0 | 0.726 | 0 | 0.82 | 0 | 0.815 | 0 |
| Dracaena reflexa | | 0.906 | 0 | 0.907 | 0 | 0.927 | 0 | 0.921 | 0 |
| Drosera aliciae | | 0.906 | 0 | 0.879 | 0 | 0.928 | 0 | 0.916 | 0 |
| Drosera binata | | 0.957 | 0 | 0.884 | 0 | 0.969 | 0 | 0.959 | 0 |
| Drosera capillaris | | 0.908 | 0 | 0.906 | 0 | 0.914 | 0 | 0.902 | 0 |
| Echeveria secunda | | 0.956 | 0 | 0.829 | 0 | 0.925 | 0 | 0.919 | 0 |
| Echinochloa polystachya | | 0.778 | 0 | 0.81 | 0 | 0.88 | 0 | 0.852 | 0 |
| Echinodorus cordifolius | | 0.842 | 0 | 0.862 | 0 | 0.895 | 0 | 0.886 | 0 |
| Echinodorus subalatus | | 0.881 | 0 | 0.863 | 0 | 0.909 | 0 | 0.901 | 0 |
| Ehretia acuminata | | 0.92 | 0 | 0.911 | 0 | 0.948 | 0 | 0.94 | 0 |
| Eichhornia azurea | | 0.839 | 0 | 0.857 | 0 | 0.851 | 0 | 0.839 | 0 |
| Elaeis guineensis | | 0.8 | 0 | 0.682 | 0 | 0.86 | 0 | 0.851 | 0 |
| Embothrium coccineum | | 0.98 | 0 | 0.907 | 0 | 0.985 | 0 | 0.977 | 0 |
| Emilia sonchifolia | | 0.771 | 0 | 0.79 | 0 | 0.867 | 0 | 0.854 | 0 |
| Empetrum rubrum | | 0.572 | 33.3 | 0.859 | 0 | 0.962 | 0 | 0.956 | 0 |
| Entada phaseoloides | | 0.905 | 0 | 0.855 | 0 | 0.936 | 0 | 0.948 | 36.7 |
| Epacris impressa | | 0.981 | 0 | 0.981 | 0 | 0.989 | 0 | 0.975 | 0 |
| Epipremnum pinnatum | | 0.758 | 0 | 0.804 | 0 | 0.821 | 0 | 0.82 | 0 |
| Eragrostis trichodes | | 0.938 | 0 | 0.92 | 0 | 0.933 | 0 | 0.924 | 0 |
| Erica glandulosa | | 0.992 | 0 | 0.949 | 0 | 0.964 | 0 | 0.962 | 0 |
| Eriochloa villosa | | 0.888 | 0 | 0.867 | 0 | 0.923 | 0 | 0.902 | 0 |
| Erythrina crista-galli | | 0.773 | 0 | 0.771 | 0 | 0.836 | 0 | 0.819 | 0 |
| Erythrina herbacea | | 0.909 | 0 | 0.934 | 0 | 0.912 | 0 | 0.9 | 0 |
| Etlingera elatior | | 0.832 | 0 | 0.782 | 0 | 0.84 | 0 | 0.828 | 0 |
| Eucalyptus cinerea | | 0.976 | 0 | 0.902 | 0 | 0.902 | 0 | 0.9 | 0 |
| Eucalyptus cladocalyx | | 0.946 | 0 | 0.928 | 0 | 0.93 | 0 | 0.924 | 0 |
| Eucalyptus comuta | | 0.935 | 0 | 0.936 | 0 | 0.965 | 0 | 0.966 | 3.3 |
| Eucalyptus dalrympleana | | 0.992 | 0 | 0.996 | 0 | 0.987 | 0 | 0.981 | 0 |
| Eucalyptus goniocalyx | | 0.97 | 0 | 0.966 | 0 | 0.986 | 0 | 0.977 | 0 |
| Eucalyptus leucoxylon | | 0.942 | 0 | 0.953 | 0 | 0.972 | 0 | 0.97 | 0 |
| Eucalyptus nitens | | 0.931 | 0 | 0.879 | 0 | 0.953 | 0 | 0.978 | 90 |
| Eucalyptus ovata | | 0.962 | 0 | 0.975 | 0 | 0.984 | 0 | 0.977 | 0 |
| Eucalyptus rubida | | 0.984 | 0 | 0.989 | 0 | 0.984 | 0 | 0.979 | 0 |
| Eugenia uniflora | | 0.746 | 0 | 0.793 | 0 | 0.863 | 0 | 0.852 | 0 |
| Eulophia alta | | 0.854 | 0 | 0.835 | 0 | 0.888 | 0 | 0.877 | 0 |
| Euonymus alatus | | 0.904 | 0 | 0.9 | 0 | 0.938 | 0 | 0.926 | 0 |
| Euonymus hamiltonianus | | 0.752 | 0 | 0.849 | 0 | 0.865 | 0 | 0.846 | 0 |
| Euphorbia balsamifera | | 0.904 | 0 | 0.844 | 0 | 0.905 | 0 | 0.898 | 0 |
| Euphorbia leucocephala | | 0.73 | 0 | 0.776 | 0 | 0.843 | 0 | 0.85 | 0 |
| Euphorbia mauritanica | | 0.949 | 0 | 0.936 | 0 | 0.967 | 0 | 0.95 | 0 |
| Euphorbia tithymaloides | | 0.78 | 0 | 0.798 | 0 | 0.831 | 0 | 0.817 | 0 |
| Euryops abrotanifolius | | 0.96 | 0 | 0.821 | 0 | 0.973 | 0 | 0.95 | 0 |
| Euryops chrysanthemoides | | 0.866 | 0 | 0.817 | 0 | 0.97 | 0 | 0.973 | 0 |

| species | model | GLM | | GAM | | RF | | GBM | |
|--------------------------------|-------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Ficus auriculata | | 0.863 | 0 | 0.715 | 0 | 0.9 | 0 | 0.885 | 0 |
| Ficus benjamina | | 0.786 | 0 | 0.801 | 0 | 0.813 | 0 | 0.809 | 0 |
| Ficus drupacea | | 0.911 | 0 | 0.819 | 0 | 0.872 | 0 | 0.859 | 0 |
| Ficus elastica | | 0.535 | 66.7 | 0.597 | 0 | 0.717 | 6.7 | 0.714 | 6.7 |
| Ficus erecta | | 0.944 | 0 | 0.939 | 0 | 0.968 | 0 | 0.96 | 0 |
| Ficus lutea | | 0.835 | 0 | 0.813 | 0 | 0.863 | 0 | 0.862 | 0 |
| Ficus palmata | | 0.664 | 0 | 0.834 | 0 | 0.843 | 0 | 0.823 | 0 |
| Ficus racemosa | | 0.885 | 0 | 0.898 | 0 | 0.919 | 0 | 0.91 | 0 |
| Ficus virens | | 0.851 | 0 | 0.86 | 0 | 0.921 | 0 | 0.885 | 0 |
| Firmiana simplex | | 0.9 | 0 | 0.872 | 0 | 0.9 | 0 | 0.872 | 0 |
| Fragaria chiloensis | | 0.859 | 0 | 0.878 | 0 | 0.911 | 0 | 0.903 | 0 |
| Frangula purshiana | | 0.97 | 0 | 0.94 | 0 | 0.96 | 0 | 0.947 | 0 |
| Fraxinus nigra | | 0.912 | 0 | 0.926 | 0 | 0.96 | 0 | 0.951 | 0 |
| Fraxinus uhdei | | 0.879 | 0 | 0.863 | 0 | 0.879 | 0 | 0.866 | 0 |
| Freesia laxa | | 0.89 | 0 | 0.913 | 0 | 0.938 | 0 | 0.92 | 0 |
| Fuchsia paniculata | | 0.922 | 0 | 0.918 | 0 | 0.935 | 0 | 0.914 | 0 |
| Galphimia glauca | | 0.888 | 0 | 0.878 | 0 | 0.891 | 0 | 0.886 | 0 |
| Galphimia gracilis | | 0.771 | 0 | 0.696 | 0 | 0.843 | 0 | 0.817 | 0 |
| Garcinia livingstonei | | 0.868 | 0 | 0.864 | 0 | 0.869 | 0 | 0.848 | 0 |
| Gardenia jasminoides | | 0.838 | 0 | 0.86 | 0 | 0.84 | 0 | 0.828 | 0 |
| Geitonoplesium cymosum | | 0.988 | 0 | 0.984 | 0 | 0.984 | 0 | 0.979 | 0 |
| Geranium incanum | | 0.906 | 0 | 0.914 | 0 | 0.928 | 0 | 0.913 | 0 |
| Geranium thunbergii | | 0.9 | 0 | 0.775 | 0 | 0.961 | 0 | 0.956 | 0 |
| Gerbera jamesonii | | 0.805 | 0 | 0.795 | 0 | 0.87 | 3.3 | 0.851 | 3.3 |
| Gibasis pellucida | | 0.838 | 0 | 0.798 | 0 | 0.865 | 0 | 0.851 | 0 |
| Gilia tricolor | | 0.902 | 0 | 0.923 | 0 | 0.948 | 0 | 0.939 | 0 |
| Gladiolus papilio | | 0.933 | 0 | 0.93 | 33.3 | 0.961 | 0 | 0.941 | 0 |
| Gladiolus tristis | | 0.866 | 0 | 0.915 | 0 | 0.947 | 0 | 0.931 | 0 |
| Gladiolus undulatus | | 0.922 | 0 | 0.925 | 0 | 0.955 | 0 | 0.942 | 0 |
| Glandularia peruviana | | 0.926 | 0 | 0.918 | 0 | 0.943 | 0 | 0.929 | 0 |
| Glandularia tenera | | 0.895 | 0 | 0.833 | 0 | 0.919 | 0 | 0.913 | 0 |
| Gloriosa superba | | 0.781 | 0 | 0.805 | 0 | 0.876 | 0 | 0.869 | 0 |
| Gloxinia perennis | | 0.855 | 0 | 0.786 | 0 | 0.905 | 0 | 0.907 | 0 |
| Gloxinia sylvatica | | 0.764 | 0 | 0.811 | 0 | 0.917 | 0 | 0.903 | 0 |
| Gmelina arborea | | 0.781 | 0 | 0.742 | 0 | 0.805 | 0 | 0.798 | 0 |
| Gomphrena globosa | | 0.687 | 0 | 0.658 | 0 | 0.664 | 3.3 | 0.651 | 3.3 |
| Grevillea banksii | | 0.706 | 33.3 | 0.861 | 0 | 0.869 | 0 | 0.859 | 0 |
| Grevillea juniperina | | 0.949 | 0 | 0.935 | 0 | 0.978 | 0 | 0.972 | 0 |
| Grevillea rosmarinifolia | | 0.926 | 0 | 0.919 | 0 | 0.98 | 0 | 0.969 | 0 |
| Hakea eriantha | | 0.978 | 0 | 0.97 | 0 | 0.978 | 0 | 0.972 | 0 |
| Hakea laurina | | 0.912 | 0 | 0.914 | 0 | 0.963 | 0 | 0.963 | 0 |
| Haloragis erecta | | 0.987 | 0 | 0.939 | 0 | 0.953 | 0 | 0.945 | 0 |
| Hamelia patens | | 0.781 | 0 | 0.81 | 0 | 0.892 | 0 | 0.872 | 0 |
| Hardenbergia comptoniana | | 0.752 | 0 | 0.868 | 0 | 0.944 | 0 | 0.93 | 0 |
| Harpephyllum caffrum | | 0.846 | 0 | 0.88 | 0 | 0.913 | 0 | 0.903 | 0 |
| Harrisia pomanensis | | 0.784 | 0 | 0.923 | 0 | 0.945 | 0 | 0.894 | 0 |
| Hebenstretia dentata | | 0.869 | 0 | 0.84 | 0 | 0.911 | 0 | 0.903 | 0 |
| Hedychium coronarium | | 0.806 | 0 | 0.811 | 0 | 0.844 | 0 | 0.835 | 0 |
| Heimia salicifolia | | 0.803 | 0 | 0.839 | 0 | 0.887 | 0 | 0.874 | 0 |
| Helanthium bolivianum | | 0.801 | 0 | 0.803 | 0 | 0.86 | 0 | 0.857 | 0 |
| Helanthium tenellum | | 0.691 | 0 | 0.721 | 0 | 0.815 | 0 | 0.79 | 0 |
| Helenium bigelovii | | 0.939 | 0 | 0.944 | 0 | 0.956 | 0 | 0.953 | 0 |
| Helianthus angustifolius | | 0.962 | 0 | 0.948 | 0 | 0.916 | 0 | 0.916 | 0 |
| Helianthus debilis | | 0.735 | 0 | 0.72 | 0 | 0.817 | 0 | 0.807 | 0 |
| Helianthus giganteus | | 0.878 | 0 | 0.817 | 0 | 0.886 | 0 | 0.875 | 0 |
| Helianthus salicifolius | | 0.925 | 0 | 0.954 | 0 | 0.987 | 0 | 0.98 | 0 |
| Herbertia lahue | | 0.915 | 0 | 0.918 | 0 | 0.927 | 0 | 0.903 | 0 |
| Hesperantha coccinea | | 0.968 | 0 | 0.9 | 0 | 0.962 | 0 | 0.95 | 0 |
| Hesperantha falcata | | 0.938 | 0 | 0.949 | 0 | 0.975 | 0 | 0.962 | 0 |
| Heterocentron subtriplinervium | | 0.917 | 0 | 0.865 | 0 | 0.942 | 0 | 0.909 | 0 |
| Heterotis rotundifolia | | 0.855 | 0 | 0.846 | 0 | 0.875 | 0 | 0.871 | 0 |
| Hevea brasiliensis | | 0.846 | 0 | 0.821 | 0 | 0.882 | 0 | 0.883 | 0 |
| Hibiscus acetosella | | 0.803 | 0 | 0.699 | 0 | 0.851 | 0 | 0.851 | 0 |
| Hibiscus diversifolius | | 0.828 | 0 | 0.82 | 0 | 0.871 | 0 | 0.869 | 0 |

| species | GLM | | GAM | | RF | | GBM | |
|------------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Hibiscus mutabilis | 0.792 | 0 | 0.721 | 0 | 0.833 | 0 | 0.828 | 0 |
| Hibiscus sabdariffa | 0.735 | 0 | 0.679 | 0 | 0.797 | 0 | 0.795 | 0 |
| Hibiscus schizopetalus | 0.753 | 0 | 0.704 | 0 | 0.774 | 3.3 | 0.769 | 0 |
| Hibiscus tiliaceus | 0.858 | 0 | 0.852 | 0 | 0.898 | 0 | 0.884 | 0 |
| Hippeastrum puniceum | 0.744 | 0 | 0.642 | 0 | 0.85 | 0 | 0.833 | 0 |
| Hippobroma longiflora | 0.842 | 0 | 0.822 | 0 | 0.881 | 0 | 0.877 | 0 |
| Hiptage benghalensis | 0.909 | 0 | 0.73 | 33.3 | 0.951 | 0 | 0.909 | 0 |
| Holmskioldia sanguinea | 0.689 | 0 | 0.761 | 0 | 0.811 | 3.3 | 0.797 | 0 |
| Homalanthus populifolius | 0.937 | 0 | 0.922 | 0 | 0.95 | 0 | 0.944 | 0 |
| Homalocladium platycladum | 0.783 | 0 | 0.703 | 0 | 0.842 | 0 | 0.818 | 0 |
| Houstonia caerulea | 0.979 | 0 | 0.955 | 0 | 0.986 | 0 | 0.972 | 0 |
| Houttuynia cordata | 0.849 | 0 | 0.833 | 0 | 0.894 | 0 | 0.885 | 0 |
| Hovea pungens | 0.976 | 0 | 0.959 | 0 | 0.986 | 0 | 0.979 | 0 |
| Hovenia dulcis | 0.829 | 0 | 0.841 | 0 | 0.923 | 0 | 0.896 | 0 |
| Hoya australis | 0.841 | 0 | 0.887 | 0 | 0.92 | 0 | 0.901 | 0 |
| Hoya carnea | 0.903 | 0 | 0.733 | 0 | 0.857 | 0 | 0.818 | 0 |
| Hura crepitans | 0.822 | 0 | 0.817 | 0 | 0.858 | 0 | 0.85 | 0 |
| Hydrangea paniculata | 0.937 | 0 | 0.915 | 0 | 0.927 | 0 | 0.919 | 0 |
| Hydrocleys nymphoides | 0.715 | 0 | 0.583 | 33.3 | 0.736 | 3.3 | 0.723 | 3.3 |
| Hydrocotyle americana | 0.93 | 0 | 0.963 | 0 | 0.955 | 0 | 0.946 | 0 |
| Hydrolea spinosa | 0.798 | 0 | 0.811 | 0 | 0.876 | 0 | 0.864 | 0 |
| Hylotelephium erythrostictum | 0.914 | 0 | 0.912 | 0 | 0.928 | 0 | 0.922 | 0 |
| Hymenocallis littoralis | 0.805 | 0 | 0.826 | 0 | 0.854 | 0 | 0.85 | 0 |
| Hypericum gramineum | 0.914 | 0 | 0.92 | 0 | 0.961 | 0 | 0.95 | 0 |
| Hypericum hypericoides | 0.912 | 0 | 0.924 | 0 | 0.95 | 0 | 0.944 | 0 |
| Hypericum patulum | 0.864 | 0 | 0.859 | 0 | 0.82 | 0 | 0.752 | 20 |
| Hypoestes aristata | 0.905 | 0 | 0.877 | 0 | 0.878 | 0 | 0.881 | 0 |
| Hypoestes phyllostachya | 0.829 | 0 | 0.864 | 0 | 0.857 | 0 | 0.837 | 0 |
| Idesia polycarpa | 0.966 | 0 | 0.836 | 0 | 0.923 | 0 | 0.923 | 0 |
| Ilex crenata | 0.938 | 0 | 0.895 | 0 | 0.928 | 0 | 0.915 | 0 |
| Ilex paraguariensis | 0.804 | 0 | 0.868 | 0 | 0.9 | 0 | 0.885 | 0 |
| Ilex rotunda | 0.931 | 0 | 0.878 | 0 | 0.948 | 0 | 0.94 | 0 |
| Indigofera tinctoria | 0.931 | 0 | 0.936 | 0 | 0.96 | 0 | 0.948 | 0 |
| Inga edulis | 0.853 | 0 | 0.849 | 0 | 0.892 | 0 | 0.886 | 0 |
| Ipomoea alba | 0.76 | 0 | 0.763 | 0 | 0.85 | 0 | 0.84 | 0 |
| Ipomoea cairica | 0.772 | 0 | 0.774 | 0 | 0.83 | 0 | 0.815 | 0 |
| Ipomoea carnea | 0.756 | 0 | 0.784 | 0 | 0.833 | 0 | 0.818 | 0 |
| Ipomoea pes caprae | 0.812 | 0 | 0.819 | 0 | 0.867 | 0 | 0.85 | 0 |
| Ipomoea tricolor | 0.67 | 0 | 0.635 | 0 | 0.795 | 0 | 0.807 | 0 |
| Iris domestica | 0.851 | 0 | 0.863 | 0 | 0.896 | 0 | 0.888 | 0 |
| Iris japonica | 0.923 | 0 | 0.921 | 0 | 0.965 | 0 | 0.954 | 0 |
| Iris sanguinea | 0.652 | 33.3 | 0.986 | 0 | 0.939 | 0 | 0.93 | 0 |
| Isotoma fluviatilis | 0.942 | 0 | 0.957 | 0 | 0.934 | 0 | 0.922 | 0 |
| Ixia polystachya | 0.939 | 0 | 0.963 | 0 | 0.973 | 0 | 0.964 | 0 |
| Ixora coccinea | 0.788 | 0 | 0.766 | 0 | 0.799 | 0 | 0.801 | 0 |
| Jasminum dichotomum | 0.877 | 0 | 0.852 | 0 | 0.876 | 0 | 0.887 | 0 |
| Jasminum fluminense | 0.808 | 0 | 0.811 | 0 | 0.864 | 0 | 0.858 | 0 |
| Jasminum grandiflorum | 0.631 | 0 | 0.57 | 33.3 | 0.72 | 3.3 | 0.697 | 3.3 |
| Jasminum multiflorum | 0.855 | 0 | 0.842 | 0 | 0.769 | 3.3 | 0.754 | 3.3 |
| Jasminum polyanthum | 0.804 | 0 | 0.843 | 0 | 0.927 | 0 | 0.918 | 0 |
| Jasminum simplicifolium | 0.94 | 0 | 0.852 | 0 | 0.939 | 0 | 0.931 | 0 |
| Justicia betonica | 0.838 | 0 | 0.821 | 0 | 0.88 | 0 | 0.874 | 0 |
| Justicia brandegeana | 0.683 | 0 | 0.907 | 0 | 0.8 | 0 | 0.772 | 0 |
| Justicia spicigera | 0.754 | 0 | 0.736 | 0 | 0.857 | 0 | 0.85 | 0 |
| Kalanchoe crenata | 0.884 | 0 | 0.749 | 33.3 | 0.841 | 0 | 0.841 | 0 |
| Kalopanax septemlobus | 0.9 | 0 | 0.87 | 0 | 0.941 | 0 | 0.936 | 0 |
| Kennedia rubicunda | 0.987 | 0 | 0.977 | 0 | 0.979 | 0 | 0.972 | 0 |
| Kigelia africana | 0.762 | 0 | 0.778 | 0 | 0.83 | 0 | 0.817 | 0 |
| Kolkwitzia amabilis | 0.913 | 0 | 0.881 | 0 | 0.921 | 0 | 0.919 | 0 |
| Kummerowia stipulacea | 0.917 | 0 | 0.901 | 0 | 0.964 | 0 | 0.96 | 0 |
| Kummerowia striata | 0.932 | 0 | 0.897 | 0 | 0.954 | 0 | 0.947 | 0 |
| Kunzea ambigua | 0.949 | 0 | 0.929 | 0 | 0.973 | 0 | 0.97 | 0 |
| Kunzea ericoides | 0.98 | 0 | 0.978 | 0 | 0.983 | 0 | 0.981 | 0 |

| species | GLM | | GAM | | RF | | GBM | |
|-----------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Lablab purpureus | 0.752 | 0 | 0.762 | 0 | 0.804 | 0 | 0.799 | 0 |
| Laelia rubescens | 0.863 | 0 | 0.927 | 0 | 0.881 | 0 | 0.856 | 0 |
| Lagerstroemia speciosa | 0.836 | 0 | 0.763 | 0 | 0.873 | 0 | 0.853 | 0 |
| Lampranthus spectabilis | 0.787 | 0 | 0.751 | 0 | 0.91 | 0 | 0.887 | 0 |
| Lawsonia inermis | 0.762 | 0 | 0.755 | 0 | 0.817 | 0 | 0.817 | 0 |
| Leonotis leonurus | NaN | 100 | 0.944 | 0 | 0.931 | 0 | 0.915 | 13.3 |
| Leonotis ocymifolia | 0.921 | 0 | 0.916 | 0 | 0.916 | 0 | 0.909 | 0 |
| Leonurus japonicus | 0.731 | 0 | 0.7 | 0 | 0.823 | 0 | 0.802 | 0 |
| Leonurus sibiricus | 0.682 | 0 | 0.705 | 0 | 0.829 | 0 | 0.809 | 0 |
| Leptospermum polygalifolium | 0.978 | 0 | 0.989 | 0 | 0.987 | 0 | 0.982 | 0 |
| Lespedeza bicolor | 0.867 | 0 | 0.814 | 0 | 0.928 | 0 | 0.911 | 0 |
| Lespedeza cyrtobotrya | 0.982 | 0 | 0.969 | 0 | 0.986 | 0 | 0.982 | 0 |
| Lespedeza thunbergii | 0.881 | 0 | 0.897 | 0 | 0.939 | 0 | 0.918 | 0 |
| Leucophyllum frutescens | 0.755 | 0 | 0.94 | 0 | 0.956 | 0 | 0.935 | 0 |
| Ligustrum obtusifolium | 0.883 | 0 | 0.889 | 0 | 0.944 | 0 | 0.935 | 0 |
| Ligustrum tschonoskii | 0.959 | 0 | 0.923 | 0 | 0.968 | 0 | 0.963 | 0 |
| Lilium formosanum | 0.95 | 0 | 0.937 | 0 | 0.946 | 0 | 0.942 | 0 |
| Limnobiium laevigatum | 0.691 | 0 | 0.795 | 0 | 0.8 | 0 | 0.788 | 0 |
| Limncharis flava | 0.753 | 0 | 0.807 | 0 | 0.84 | 0 | 0.833 | 0 |
| Linaria maroccana | 0.814 | 0 | 0.832 | 0 | 0.865 | 0 | 0.837 | 0 |
| Linum grandiflorum | 0.891 | 0 | 0.894 | 0 | 0.923 | 0 | 0.911 | 0 |
| Liriope muscari | 0.884 | 0 | 0.815 | 0 | 0.911 | 0 | 0.898 | 0 |
| Liriope spicata | 0.877 | 0 | 0.863 | 0 | 0.915 | 0 | 0.917 | 0 |
| Livistona australis | 0.938 | 0 | 0.932 | 0 | 0.961 | 0 | 0.958 | 0 |
| Lobelia cardinalis | 0.727 | 0 | 0.786 | 0 | 0.876 | 0 | 0.812 | 0 |
| Lobelia inflata | 0.928 | 0 | 0.93 | 0 | 0.959 | 0 | 0.952 | 0 |
| Lonicera sempervirens | 0.96 | 0 | 0.937 | 0 | 0.957 | 0 | 0.935 | 0 |
| Lophostemon confertus | 0.953 | 0 | 0.936 | 0 | 0.963 | 0 | 0.957 | 0 |
| Ludwigia alternifolia | 0.962 | 0 | 0.958 | 0 | 0.979 | 0 | 0.977 | 0 |
| Ludwigia octovalvis | 0.724 | 0 | 0.76 | 0 | 0.865 | 0 | 0.836 | 0 |
| Ludwigia peruviana | 0.776 | 0 | 0.79 | 0 | 0.848 | 0 | 0.837 | 0 |
| Luffa cylindrica | 0.697 | 0 | 0.669 | 0 | 0.794 | 0 | 0.794 | 0 |
| Luma apiculata | 0.841 | 0 | 0.842 | 0 | 0.91 | 0 | 0.902 | 0 |
| Lupinus mexicanus | 0.95 | 0 | 0.891 | 0 | 0.947 | 0 | 0.94 | 0 |
| Lycianthes rantonnetii | 0.638 | 0 | 0.66 | 33.3 | 0.73 | 0 | 0.708 | 3.3 |
| Lycoris radiata | 0.917 | 0 | 0.891 | 0 | 0.993 | 0 | 0.963 | 0 |
| Lysimachia japonica | 0.945 | 0 | 0.974 | 0 | 0.958 | 0 | 0.946 | 0 |
| Maackia amurensis | 0.968 | 0 | 0.926 | 0 | 0.923 | 0 | 0.918 | 0 |
| Magnolia grandiflora | 0.744 | 0 | 0.722 | 0 | 0.789 | 0 | 0.777 | 0 |
| Magnolia kobus | 0.586 | 33.3 | 0.681 | 0 | 0.881 | 0 | 0.869 | 0 |
| Magnolia obovata | 0.866 | 0 | 0.833 | 0 | 0.93 | 0 | 0.924 | 0 |
| Malephora crocea | 0.924 | 0 | 0.946 | 0 | 0.964 | 0 | 0.958 | 0 |
| Mallotus philippensis | 0.783 | 0 | 0.799 | 0 | 0.868 | 0 | 0.847 | 0 |
| Malpighia emarginata | 0.8 | 0 | 0.776 | 0 | 0.846 | 0 | 0.833 | 0 |
| Malpighia glabra | 0.779 | 0 | 0.805 | 0 | 0.867 | 0 | 0.851 | 0 |
| Malpighia mexicana | 0.874 | 0 | 0.841 | 0 | 0.945 | 0 | 0.933 | 0 |
| Malus prunifolia | 0.876 | 0 | 0.879 | 0 | 0.9 | 0 | 0.889 | 0 |
| Malva assurgentiflora | 0.954 | 0 | 0.917 | 0 | 0.957 | 0 | 0.942 | 0 |
| Malvaviscus arboreus | 0.788 | 0 | 0.809 | 0 | 0.886 | 0 | 0.85 | 0 |
| Mammea americana | 0.852 | 0 | 0.825 | 0 | 0.813 | 0 | 0.802 | 0 |
| Mandevilla laxa | 0.823 | 0 | 0.832 | 0 | 0.933 | 0 | 0.924 | 0 |
| Mangifera indica | 0.743 | 0 | 0.769 | 0 | 0.825 | 0 | 0.807 | 0 |
| Manilkara zapota | 0.815 | 0 | 0.839 | 0 | 0.867 | 0 | 0.863 | 0 |
| Maranta arundinacea | 0.838 | 0 | 0.856 | 0 | 0.868 | 0 | 0.868 | 0 |
| Margyricarpus pinnatus | 0.877 | 0 | 0.81 | 0 | 0.917 | 0 | 0.909 | 0 |
| Martynia annua | 0.789 | 0 | 0.804 | 0 | 0.844 | 0 | 0.841 | 0 |
| Maurandya antirrhiniflora | 0.827 | 0 | 0.849 | 0 | 0.899 | 0 | 0.889 | 0 |
| Mazus pumilus | 0.847 | 0 | 0.863 | 0 | 0.903 | 0 | 0.902 | 0 |
| Melaleuca hypericifolia | 0.976 | 0 | 0.906 | 0 | 0.978 | 0 | 0.969 | 0 |
| Melastoma malabathricum | 0.799 | 0 | 0.858 | 0 | 0.901 | 0 | 0.874 | 0 |
| Meliantus major | 0.915 | 0 | 0.81 | 0 | 0.93 | 0 | 0.927 | 0 |
| Melinis repens | 0.679 | 0 | 0.708 | 0 | 0.86 | 0 | 0.812 | 0 |
| Melothria pendula | 0.729 | 0 | 0.773 | 0 | 0.86 | 0 | 0.831 | 0 |
| Miconia calvescens | 0.849 | 0 | 0.856 | 0 | 0.888 | 0 | 0.883 | 0 |

| species | model | GLM | | GAM | | RF | | GBM | |
|------------------------------|-------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Micranthemum umbrosum | | 0.731 | 0 | 0.764 | 0 | 0.846 | 0 | 0.82 | 0 |
| Mimosa pigra | | 0.762 | 0 | 0.773 | 0 | 0.875 | 0 | 0.861 | 0 |
| Mimusops elengi | | 0.888 | 0 | 0.867 | 0 | 0.904 | 0 | 0.891 | 60 |
| Molineria capitulata | | 0.774 | 0 | 0.66 | 0 | 0.797 | 0 | 0.792 | 0 |
| Momordica balsamina | | 0.75 | 0 | 0.775 | 0 | 0.838 | 0 | 0.821 | 0 |
| Momordica charantia | | 0.742 | 0 | 0.751 | 0 | 0.841 | 0 | 0.829 | 0 |
| Momordica cochinchinensis | | 0.908 | 0 | 0.801 | 0 | 0.907 | 0 | 0.883 | 0 |
| Monarda fistulosa | | 0.824 | 0 | 0.863 | 0 | 0.91 | 0 | 0.879 | 0 |
| Monarda punctata | | 0.918 | 0 | 0.926 | 0 | 0.947 | 0 | 0.946 | 0 |
| Monochoria vaginalis | | 0.868 | 0 | 0.843 | 0 | 0.9 | 0 | 0.884 | 0 |
| Moraea flaccida | | NaN | 100 | 0.947 | 0 | 0.969 | 0 | 0.953 | 0 |
| Moraea fugax | | 0.942 | 0 | 0.925 | 0 | 0.975 | 0 | 0.949 | 0 |
| Moraea miniata | | NaN | 100 | 0.922 | 0 | 0.951 | 0 | 0.948 | 20 |
| Moraea polystachya | | 0.968 | 0 | 0.947 | 0 | 0.94 | 0 | 0.923 | 0 |
| Moraea setifolia | | 0.954 | 0 | 0.948 | 0 | 0.969 | 0 | 0.963 | 0 |
| Morinda citrifolia | | 0.877 | 0 | 0.885 | 0 | 0.895 | 0 | 0.89 | 0 |
| Moringa oleifera | | 0.769 | 0 | 0.746 | 0 | 0.806 | 0 | 0.813 | 0 |
| Mucuna pruriens | | 0.791 | 0 | 0.813 | 0 | 0.848 | 0 | 0.848 | 0 |
| Muehlenbeckia axillaris | | 0.994 | 0 | 0.983 | 0 | 0.984 | 0 | 0.978 | 0 |
| Mukia maderaspatana | | 0.667 | 0 | 0.748 | 0 | 0.857 | 0 | 0.838 | 0 |
| Musa acuminata | | 0.57 | 0 | 0.831 | 0 | 0.789 | 0 | 0.789 | 3.3 |
| Myrica rubra | | 0.961 | 0 | 0.943 | 0 | 0.964 | 0 | 0.96 | 0 |
| Myrmecophila tibicinis | | 0.949 | 0 | 0.936 | 0 | 0.939 | 0 | 0.919 | 0 |
| Nandina domestica | | 0.855 | 0 | 0.832 | 0 | 0.884 | 0 | 0.864 | 0 |
| Nepeta racemosa | | 0.851 | 0 | 0.902 | 0 | 0.925 | 0 | 0.918 | 0 |
| Neptunia oleracea | | 0.833 | 0 | 0.847 | 0 | 0.845 | 0 | 0.833 | 0 |
| Neptunia plena | | 0.831 | 0 | 0.822 | 0 | 0.838 | 0 | 0.829 | 0 |
| Nertera granadensis | | 0.927 | 0 | 0.911 | 0 | 0.949 | 0 | 0.944 | 0 |
| Nicotiana acuminata | | 0.915 | 0 | 0.906 | 0 | 0.928 | 0 | 0.915 | 0 |
| Nicotiana longiflora | | 0.644 | 0 | 0.677 | 0 | 0.819 | 0 | 0.796 | 0 |
| Nicotiana glauca | | 0.718 | 0 | 0.743 | 0 | 0.847 | 0 | 0.833 | 0 |
| Nothoscordum bivalve | | 0.846 | 0 | 0.89 | 0 | 0.891 | 0 | 0.869 | 0 |
| Nymphaea nouchali | | 0.86 | 0 | 0.852 | 0 | 0.889 | 0 | 0.875 | 0 |
| Nymphaea odorata | | 0.778 | 0 | 0.822 | 0 | 0.879 | 0 | 0.839 | 0 |
| Ochroma pyramidale | | 0.844 | 0 | 0.836 | 0 | 0.88 | 0 | 0.872 | 0 |
| Ocimum americanum | | 0.734 | 0 | 0.758 | 0 | 0.812 | 0 | 0.793 | 0 |
| Ocimum gratissimum | | 0.775 | 0 | 0.762 | 0 | 0.845 | 0 | 0.837 | 0 |
| Ocimum kilimandscharicum | | 0.891 | 0 | 0.844 | 0 | 0.887 | 0 | 0.871 | 0 |
| Ocimum tenuiflorum | | 0.821 | 0 | 0.815 | 0 | 0.865 | 0 | 0.854 | 0 |
| Odontonema tubaeforme | | 0.817 | 0 | 0.813 | 0 | 0.86 | 0 | 0.861 | 0 |
| Oeceoclades maculata | | 0.804 | 0 | 0.767 | 0 | 0.845 | 0 | 0.837 | 0 |
| Oenothera drummondii | | 0.63 | 0 | 0.865 | 0 | 0.899 | 0 | 0.897 | 0 |
| Oenothera perennis | | 0.951 | 0 | 0.915 | 0 | 0.946 | 0 | 0.928 | 0 |
| Opuntia aurantiaca | | 0.959 | 0 | 0.944 | 0 | 0.978 | 0 | 0.969 | 0 |
| Opuntia basilaris | | 0.777 | 0 | 0.912 | 0 | 0.948 | 0 | 0.926 | 0 |
| Opuntia humifusa | | 0.92 | 0 | 0.941 | 0 | 0.916 | 0 | 0.911 | 0 |
| Opuntia leucotricha | | 0.91 | 0 | 0.805 | 0 | 0.948 | 0 | 0.939 | 0 |
| Opuntia polyacantha | | 0.911 | 0 | 0.923 | 0 | 0.927 | 0 | 0.917 | 0 |
| Ornithogalum thyrsoides | | 0.877 | 0 | 0.84 | 0 | 0.933 | 0 | 0.909 | 0 |
| Orthrosanthus chimboracensis | | 0.917 | 0 | 0.888 | 0 | 0.96 | 0 | 0.948 | 0 |
| Osmanthus heterophyllus | | 0.898 | 0 | 0.907 | 0 | 0.937 | 0 | 0.92 | 0 |
| Osteospermum ecklonis | | 0.896 | 0 | 0.881 | 0 | 0.922 | 0 | 0.911 | 0 |
| Oxalis depressa | | 0.939 | 0 | 0.933 | 0 | 0.949 | 0 | 0.932 | 0 |
| Oxalis spiralis | | 0.932 | 0 | 0.91 | 0 | 0.943 | 0 | 0.943 | 0 |
| Oxalis tuberosa | | 0.915 | 0 | 0.857 | 0 | 0.93 | 0 | 0.933 | 0 |
| Pachira aquatica | | 0.845 | 0 | 0.854 | 0 | 0.893 | 0 | 0.883 | 0 |
| Paonia lactiflora | | 0.854 | 0 | 0.855 | 33.3 | 0.908 | 0 | 0.886 | 0 |
| Pandanus tectorius | | 0.918 | 0 | 0.933 | 0 | 0.889 | 0 | 0.888 | 0 |
| Pandorea jasminoides | | 0.843 | 0 | 0.829 | 0 | 0.955 | 0 | 0.949 | 0 |
| Pandorea pandorana | | 0.952 | 0 | 0.958 | 0 | 0.975 | 0 | 0.964 | 0 |
| Papaver aculeatum | | 0.933 | 0 | 0.922 | 0 | 0.957 | 0 | 0.954 | 0 |
| Papaver nudicaule | | 0.779 | 0 | 0.839 | 0 | 0.91 | 0 | 0.896 | 0 |
| Papaver orientale | | 0.894 | 0 | 0.908 | 0 | 0.942 | 0 | 0.926 | 0 |
| Parmentiera aculeata | | 0.852 | 0 | 0.859 | 0 | 0.893 | 0 | 0.888 | 0 |

| species | GLM | | GAM | | RF | | GBM | |
|------------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Parochetus communis | 0.888 | 0 | 0.897 | 0 | 0.906 | 0 | 0.892 | 0 |
| Passiflora amethystina | 0.869 | 0 | 0.858 | 0 | 0.938 | 0 | 0.931 | 0 |
| Passiflora coccinea | 0.853 | 0 | 0.852 | 0 | 0.904 | 0 | 0.905 | 0 |
| Passiflora foetida | 0.734 | 0 | 0.756 | 0 | 0.861 | 0 | 0.832 | 0 |
| Passiflora laurifolia | 0.878 | 0 | 0.833 | 0 | 0.931 | 0 | 0.933 | 0 |
| Passiflora ligularis | 0.836 | 0 | 0.814 | 0 | 0.91 | 0 | 0.904 | 0 |
| Passiflora mixta | 0.96 | 0 | 0.949 | 0 | 0.978 | 0 | 0.975 | 0 |
| Passiflora quadrangularis | 0.861 | 0 | 0.848 | 0 | 0.862 | 0 | 0.844 | 0 |
| Passiflora vitifolia | 0.889 | 0 | 0.865 | 0 | 0.919 | 0 | 0.914 | 0 |
| Pavonia hastata | 0.91 | 0 | 0.894 | 0 | 0.963 | 0 | 0.948 | 0 |
| Pelargonium alchemilloides | 0.967 | 0 | 0.965 | 0 | 0.941 | 0 | 0.936 | 0 |
| Pelargonium grossularioides | 0.882 | 0 | 0.88 | 0 | 0.931 | 0 | 0.927 | 0 |
| Pennisetum alopecuroides | 0.858 | 0 | 0.878 | 0 | 0.922 | 0 | 0.912 | 0 |
| Pennisetum orientale | 0.774 | 0 | 0.776 | 0 | 0.817 | 0 | 0.815 | 0 |
| Penstemon gentianoides | 0.961 | 0 | 0.872 | 0 | 0.936 | 0 | 0.93 | 0 |
| Pentas lanceolata | 0.81 | 0 | 0.834 | 0 | 0.832 | 0 | 0.829 | 0 |
| Peperomia obtusifolia | 0.816 | 0 | 0.84 | 0 | 0.893 | 0 | 0.892 | 0 |
| Pereskia aculeata | 0.885 | 0 | 0.875 | 0 | 0.902 | 0 | 0.884 | 0 |
| Perovskia atriplicifolia | 0.551 | 33.3 | 0.636 | 0 | 0.794 | 10 | 0.788 | 6.7 |
| Persea americana | 0.711 | 0 | 0.734 | 0 | 0.833 | 0 | 0.799 | 0 |
| Petrea volubilis | 0.803 | 0 | 0.825 | 0 | 0.871 | 0 | 0.866 | 26.7 |
| Petunia axillaris | 0.802 | 0 | 0.788 | 0 | 0.866 | 0 | 0.848 | 0 |
| Petunia integrifolia | 0.928 | 0 | 0.945 | 0 | 0.89 | 0 | 0.875 | 0 |
| Philadelphus mexicanus | 0.903 | 0 | 0.826 | 0 | 0.945 | 0 | 0.939 | 0 |
| Philadelphus pubescens | 0.86 | 0 | 0.9 | 0 | 0.925 | 0 | 0.92 | 0 |
| Philodendron bipinnatifidum | 0.836 | 0 | 0.785 | 0 | 0.856 | 0 | 0.833 | 0 |
| Philodendron ornatum | 0.894 | 0 | 0.885 | 0 | 0.93 | 0 | 0.926 | 0 |
| Phlox drummondii | 0.823 | 0 | 0.814 | 0 | 0.836 | 0 | 0.833 | 0 |
| Phoenix reclinata | 0.777 | 0 | 0.771 | 0 | 0.855 | 0 | 0.854 | 0 |
| Photinia glabra | 0.882 | 0 | 0.873 | 0 | 0.889 | 0 | 0.883 | 0 |
| Phyllanthus amarus | 0.794 | 0 | 0.798 | 0 | 0.857 | 0 | 0.847 | 63.3 |
| Phyllanthus emblica | 0.787 | 0 | 0.83 | 0 | 0.886 | 0 | 0.872 | 0 |
| Pimenta dioica | 0.83 | 0 | 0.813 | 0 | 0.872 | 0 | 0.863 | 0 |
| Pinellia ternata | 0.92 | 0 | 0.804 | 0 | 0.933 | 0 | 0.913 | 0 |
| Piper aduncum | 0.825 | 0 | 0.816 | 0 | 0.896 | 0 | 0.889 | 0 |
| Piscidia piscipula | 0.876 | 0 | 0.872 | 0 | 0.892 | 0 | 0.896 | 0 |
| Pistacia chinensis | 0.77 | 0 | 0.775 | 0 | 0.852 | 0 | 0.827 | 0 |
| Pithecellobium dulce | 0.732 | 0 | 0.766 | 0 | 0.818 | 0 | 0.802 | 0 |
| Pittosporum bicolor | 0.963 | 0 | 0.963 | 0 | 0.982 | 0 | 0.972 | 0 |
| Pittosporum eugenoides | 0.965 | 0 | 0.928 | 0 | 0.988 | 0 | 0.983 | 0 |
| Plantago rugelii | 0.956 | 0 | 0.977 | 0 | 0.973 | 0 | 0.97 | 33.3 |
| Plectranthus amboinicus | 0.777 | 0 | 0.799 | 0 | 0.854 | 0 | 0.839 | 0 |
| Plectranthus ciliatus | 0.989 | 0 | 0.954 | 0 | 0.976 | 0 | 0.959 | 0 |
| Plectranthus scutellarioides | 0.836 | 0 | 0.863 | 0 | 0.892 | 0 | 0.877 | 0 |
| Plectranthus verticillatus | 0.882 | 0 | 0.876 | 0 | 0.897 | 0 | 0.876 | 0 |
| Plumbago zeylanica | 0.695 | 0 | 0.73 | 0 | 0.846 | 0 | 0.816 | 0 |
| Plumeria obtusa | 0.93 | 0 | 0.952 | 0 | 0.932 | 0 | 0.917 | 0 |
| Plumeria rubra | 0.727 | 0 | 0.76 | 0 | 0.814 | 0 | 0.795 | 0 |
| Polygala lancifolia | 0.864 | 0 | 0.894 | 0 | 0.983 | 0 | 0.952 | 0 |
| Polygala senega | 0.711 | 33.3 | 0.938 | 0 | 0.9 | 0 | 0.891 | 0 |
| Pomaderris lanigera | 0.971 | 0 | 0.963 | 0 | 0.981 | 0 | 0.978 | 0 |
| Populus acuminata | 0.703 | 33.3 | 0.831 | 0 | 0.987 | 0 | 0.969 | 0 |
| Portulacaria afra | 0.955 | 0 | 0.987 | 0 | 0.93 | 0 | 0.903 | 0 |
| Pouteria caimito | 0.877 | 0 | 0.875 | 0 | 0.922 | 0 | 0.93 | 0 |
| Pratia repens | 0.999 | 0 | 0.944 | 0 | 0.975 | 0 | 0.958 | 0 |
| Prosopis chilensis | 0.649 | 0 | 0.749 | 0 | 0.864 | 0 | 0.846 | 0 |
| Prosopis juliflora | 0.624 | 0 | 0.651 | 0 | 0.766 | 0 | 0.761 | 0 |
| Prunus munsoniana | 0.895 | 0 | 0.874 | 0 | 0.97 | 0 | 0.97 | 0 |
| Prunus pumila | 0.902 | 0 | 0.928 | 0 | 0.928 | 0 | 0.921 | 0 |
| Prunus salicina | 0.881 | 0 | 0.798 | 0 | 0.887 | 0 | 0.881 | 0 |
| Psoralea pinnata | 0.89 | 0 | 0.87 | 0 | 0.935 | 0 | 0.933 | 0 |
| Pueraria montana | 0.814 | 0 | 0.811 | 0 | 0.862 | 0 | 0.851 | 0 |
| Pyracantha fortuneana | 0.897 | 0 | 0.935 | 0 | 0.884 | 0 | 0.87 | 0 |
| Pyracantha koidzumii | 0.644 | 33.3 | 0.645 | 0 | 0.841 | 0 | 0.807 | 0 |
| Pyrus calleryana | 0.818 | 0 | 0.771 | 0 | 0.85 | 0 | 0.835 | 0 |

| species | GLM | | GAM | | RF | | GBM | |
|------------------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| <i>Pyrus pyrifolia</i> | 0.885 | 0 | 0.846 | 0 | 0.892 | 0 | 0.873 | 0 |
| <i>Quassia amara</i> | 0.834 | 0 | 0.836 | 0 | 0.88 | 0 | 0.876 | 0 |
| <i>Quercus acutissima</i> | 0.875 | 0 | 0.873 | 0 | 0.917 | 0 | 0.909 | 0 |
| <i>Rauvolfia tetraphylla</i> | 0.83 | 0 | 0.853 | 0 | 0.87 | 0 | 0.859 | 0 |
| <i>Rauvolfia vomitoria</i> | 0.864 | 0 | 0.883 | 0 | 0.929 | 0 | 0.917 | 0 |
| <i>Rhamnus japonica</i> | 0.939 | 0 | 0.921 | 0 | 0.976 | 0 | 0.976 | 0 |
| <i>Rhaphiolepis indica</i> | 0.898 | 0 | 0.823 | 0 | 0.915 | 0 | 0.91 | 0 |
| <i>Rheum rhabarbarum</i> | 0.904 | 0 | 0.916 | 0 | 0.95 | 0 | 0.943 | 0 |
| <i>Rhipsalis baccifera</i> | 0.828 | 0 | 0.832 | 0 | 0.905 | 0 | 0.894 | 0 |
| <i>Rhodanthe chlorocephala</i> | 0.892 | 0 | 0.916 | 0 | 0.932 | 0 | 0.922 | 0 |
| <i>Rhododendron japonicum</i> | 0.737 | 0 | 0.769 | 0 | 0.942 | 0 | 0.925 | 0 |
| <i>Romneya coulteri</i> | 0.907 | 0 | 0.873 | 0 | 0.913 | 0 | 0.903 | 0 |
| <i>Romulea flava</i> | 0.959 | 0 | 0.958 | 0 | 0.949 | 0 | 0.936 | 0 |
| <i>Rosa banksiae</i> | 0.902 | 0 | 0.847 | 0 | 0.92 | 0 | 0.89 | 0 |
| <i>Rosa chinensis</i> | 0.832 | 0 | 0.758 | 0 | 0.819 | 0 | 0.767 | 0 |
| <i>Rosenbergiodendron formosum</i> | 0.855 | 0 | 0.815 | 0 | 0.918 | 0 | 0.918 | 0 |
| <i>Rubus ellipticus</i> | 0.909 | 0 | 0.88 | 0 | 0.905 | 0 | 0.889 | 0 |
| <i>Rubus rosifolius</i> | 0.875 | 0 | 0.903 | 0 | 0.934 | 0 | 0.931 | 0 |
| <i>Rudbeckia triloba</i> | 0.944 | 0 | 0.941 | 33.3 | 0.976 | 0 | 0.971 | 0 |
| <i>Ruellia brevifolia</i> | 0.802 | 0 | 0.828 | 0 | 0.891 | 0 | 0.868 | 0 |
| <i>Ruellia tuberosa</i> | 0.827 | 0 | 0.845 | 0 | 0.815 | 0 | 0.809 | 0 |
| <i>Russelia equisetiformis</i> | 0.661 | 0 | 0.751 | 0 | 0.853 | 0 | 0.817 | 0 |
| <i>Russelia sarmentosa</i> | 0.853 | 0 | 0.874 | 0 | 0.891 | 0 | 0.882 | 0 |
| <i>Sageretia thea</i> | 0.702 | 0 | 0.818 | 0 | 0.841 | 0 | 0.835 | 0 |
| <i>Sagittaria graminea</i> | 0.856 | 0 | 0.888 | 0 | 0.929 | 0 | 0.921 | 0 |
| <i>Sagittaria montevidensis</i> | 0.8 | 0 | 0.791 | 0 | 0.875 | 0 | 0.856 | 0 |
| <i>Salix gracilistyla</i> | 0.957 | 0 | 0.945 | 0 | 0.991 | 0 | 0.988 | 0 |
| <i>Salix humboldtiana</i> | 0.741 | 0 | 0.777 | 0 | 0.855 | 0 | 0.835 | 0 |
| <i>Salix nigra</i> | 0.901 | 0 | 0.896 | 0 | 0.915 | 0 | 0.901 | 0 |
| <i>Salvia africana lutea</i> | 0.946 | 0 | 0.935 | 0 | 0.997 | 0 | 0.985 | 0 |
| <i>Salvia leucantha</i> | 0.651 | 33.3 | 0.688 | 0 | 0.813 | 0 | 0.802 | 0 |
| <i>Salvia leucophylla</i> | 0.948 | 0 | 0.901 | 0 | 0.925 | 0 | 0.908 | 0 |
| <i>Salvia microphylla</i> | 0.875 | 0 | 0.881 | 0 | 0.908 | 0 | 0.902 | 0 |
| <i>Salvia plebeia</i> | 0.829 | 0 | 0.8 | 0 | 0.851 | 0 | 0.837 | 0 |
| <i>Salvia splendens</i> | 0.516 | 66.7 | 0.569 | 33.3 | 0.782 | 0 | 0.751 | 0 |
| <i>Sansevieria hyacinthoides</i> | 0.818 | 0 | 0.869 | 0 | 0.867 | 0 | 0.857 | 0 |
| <i>Santalum album</i> | 0.862 | 0 | 0.839 | 0 | 0.888 | 0 | 0.891 | 0 |
| <i>Sanvitalia procumbens</i> | 0.772 | 0 | 0.889 | 0 | 0.909 | 0 | 0.888 | 0 |
| <i>Sauropus androgynus</i> | 0.832 | 0 | 0.834 | 0 | 0.877 | 0 | 0.871 | 0 |
| <i>Scadoxus multiflorus</i> | 0.778 | 0 | 0.803 | 0 | 0.851 | 0 | 0.84 | 0 |
| <i>Scaevola taccada</i> | 0.885 | 0 | 0.896 | 0 | 0.906 | 0 | 0.903 | 0 |
| <i>Schefflera arboricola</i> | 0.89 | 0 | 0.83 | 0 | 0.933 | 0 | 0.918 | 0 |
| <i>Schisandra chinensis</i> | 0.71 | 0 | 0.77 | 0 | 0.951 | 0 | 0.945 | 0 |
| <i>Schotia brachypetala</i> | 0.951 | 0 | 0.949 | 0 | 0.975 | 0 | 0.955 | 0 |
| <i>Senecio radicans</i> | 0.943 | 0 | 0.961 | 0 | 0.961 | 0 | 0.942 | 0 |
| <i>Senecio tamoides</i> | 0.871 | 0 | 0.944 | 0 | 0.969 | 0 | 0.956 | 0 |
| <i>Senna artemisioides</i> | 0.927 | 0 | 0.943 | 0 | 0.976 | 0 | 0.967 | 0 |
| <i>Senna italica</i> | 0.806 | 0 | 0.846 | 0 | 0.899 | 0 | 0.88 | 0 |
| <i>Senna siamea</i> | 0.792 | 0 | 0.785 | 0 | 0.831 | 0 | 0.82 | 0 |
| <i>Sesbania grandiflora</i> | 0.853 | 0 | 0.712 | 0 | 0.842 | 0 | 0.825 | 0 |
| <i>Sesbania sesban</i> | 0.762 | 0 | 0.759 | 0 | 0.867 | 0 | 0.855 | 0 |
| <i>Sisyrinchium atlanticum</i> | 0.893 | 0 | 0.946 | 0 | 0.946 | 0 | 0.934 | 0 |
| <i>Sisyrinchium micranthum</i> | 0.867 | 0 | 0.893 | 0 | 0.91 | 0 | 0.906 | 0 |
| <i>Sisyrinchium mucronatum</i> | 0.916 | 0 | 0.869 | 0 | 0.948 | 0 | 0.924 | 0 |
| <i>Solandra maxima</i> | 0.85 | 0 | 0.763 | 33.3 | 0.818 | 0 | 0.791 | 0 |
| <i>Solanum aviculare</i> | 0.91 | 0 | 0.933 | 0 | 0.949 | 0 | 0.936 | 0 |
| <i>Solanum betaceum</i> | 0.873 | 0 | 0.782 | 0 | 0.826 | 0 | 0.836 | 0 |
| <i>Solanum capsicoides</i> | 0.802 | 0 | 0.794 | 0 | 0.855 | 0 | 0.85 | 0 |
| <i>Solanum chrysotrichum</i> | 0.873 | 0 | 0.625 | 33.3 | 0.92 | 0 | 0.9 | 0 |
| <i>Solanum lanceifolium</i> | 0.832 | 0 | 0.827 | 0 | 0.87 | 0 | 0.869 | 0 |
| <i>Solanum mammosum</i> | 0.828 | 0 | 0.814 | 0 | 0.845 | 0 | 0.834 | 0 |
| <i>Solanum quitoense</i> | 0.925 | 0 | 0.885 | 0 | 0.9 | 0 | 0.91 | 0 |
| <i>Solanum retroflexum</i> | 0.925 | 0 | 0.901 | 0 | 0.935 | 0 | 0.924 | 0 |
| <i>Solanum seaforthianum</i> | 0.757 | 0 | 0.789 | 0 | 0.846 | 0 | 0.834 | 0 |

| species | GLM | | GAM | | RF | | GBM | |
|-----------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Solanum sessiliflorum | 0.87 | 0 | 0.858 | 0 | 0.925 | 0 | 0.918 | 0 |
| Solanum wendlandii | 0.813 | 0 | 0.878 | 0 | 0.898 | 0 | 0.857 | 0 |
| Solidago altissima | 0.801 | 0 | 0.824 | 0 | 0.885 | 0 | 0.864 | 0 |
| Solidago ptarmicoides | 0.731 | 33.3 | 0.847 | 0 | 0.857 | 0 | 0.857 | 0 |
| Sophora tetraptera | 0.888 | 0 | 0.887 | 0 | 0.956 | 0 | 0.947 | 0 |
| Sorbus alnifolia | 0.932 | 0 | 0.843 | 0 | 0.929 | 0 | 0.926 | 0 |
| Sparaxis grandiflora | 0.94 | 0 | 0.965 | 0 | 0.987 | 0 | 0.973 | 0 |
| Spathiphyllum cannifolium | 0.903 | 0 | 0.904 | 0 | 0.929 | 0 | 0.932 | 0 |
| Spathodea campanulata | 0.803 | 0 | 0.781 | 0 | 0.836 | 0 | 0.836 | 0 |
| Spathoglottis plicata | 0.898 | 0 | 0.709 | 0 | 0.91 | 0 | 0.91 | 0 |
| Spiraea prunifolia | 0.802 | 0 | 0.84 | 0 | 0.886 | 0 | 0.874 | 0 |
| Spondias dulcis | 0.795 | 0 | 0.724 | 0 | 0.881 | 0 | 0.857 | 0 |
| Spondias mombin | 0.829 | 0 | 0.824 | 0 | 0.887 | 0 | 0.877 | 0 |
| Stachytarpheta jamaicensis | 0.837 | 0 | 0.851 | 0 | 0.884 | 0 | 0.881 | 6.7 |
| Stachytarpheta mutabilis | 0.806 | 0 | 0.679 | 33.3 | 0.819 | 0 | 0.809 | 0 |
| Stapelia grandiflora | 0.953 | 0 | 0.946 | 0 | 0.948 | 0 | 0.929 | 3.3 |
| Stenocarpus sinuatus | 0.685 | 0 | 0.857 | 0 | 0.895 | 0 | 0.869 | 0 |
| Sterculia apetala | 0.819 | 0 | 0.837 | 0 | 0.897 | 0 | 0.887 | 0 |
| Stigmaphyllon ellipticum | 0.891 | 0 | 0.877 | 0 | 0.875 | 0 | 0.874 | 0 |
| Stipa tenuissima | 0.875 | 0 | 0.85 | 0 | 0.886 | 0 | 0.881 | 0 |
| Streptosolen jamesonii | 0.903 | 0 | 0.8 | 0 | 0.927 | 0 | 0.881 | 13.3 |
| Strophanthus gratus | 0.894 | 0 | 0.911 | 0 | 0.933 | 0 | 0.929 | 0 |
| Strophanthus preussii | 0.819 | 0 | 0.753 | 0 | 0.902 | 0 | 0.898 | 0 |
| Styrax japonicus | 0.958 | 0 | 0.865 | 0 | 0.968 | 0 | 0.953 | 0 |
| Syagrus romanzoffiana | 0.867 | 0 | 0.848 | 0 | 0.914 | 0 | 0.889 | 0 |
| Symplocos paniculata | 0.903 | 0 | 0.806 | 0 | 0.95 | 0 | 0.935 | 0 |
| Syngonium angustatum | 0.905 | 0 | 0.857 | 0 | 0.921 | 0 | 0.92 | 0 |
| Syngonium podophyllum | 0.827 | 0 | 0.849 | 0 | 0.899 | 0 | 0.894 | 0 |
| Syringa reticulata | 0.703 | 0 | 0.849 | 0 | 0.906 | 0 | 0.881 | 0 |
| Syzygium paniculatum | 0.863 | 0 | 0.839 | 0 | 0.931 | 0 | 0.924 | 0 |
| Tabebuia aurea | 0.742 | 0 | 0.742 | 0 | 0.88 | 0 | 0.869 | 0 |
| Tabernaemontana divaricata | 0.8 | 0 | 0.787 | 0 | 0.817 | 0 | 0.803 | 0 |
| Tacca leontopetaloides | 0.874 | 0 | 0.872 | 0 | 0.911 | 0 | 0.903 | 0 |
| Tagetes lucida | 0.911 | 0 | 0.915 | 0 | 0.913 | 0 | 0.903 | 0 |
| Tagetes tenuifolia | 0.858 | 0 | 0.921 | 0 | 0.883 | 0 | 0.857 | 0 |
| Tamarindus indica | 0.77 | 0 | 0.793 | 0 | 0.843 | 0 | 0.833 | 0 |
| Tamarix aphylla | 0.804 | 0 | 0.807 | 0 | 0.828 | 0 | 0.812 | 0 |
| Tamarix chinensis | 0.748 | 0 | 0.732 | 0 | 0.822 | 0 | 0.806 | 0 |
| Tanacetum coccineum | 0.835 | 0 | 0.809 | 0 | 0.939 | 0 | 0.933 | 0 |
| Tecoma stans | 0.671 | 0 | 0.7 | 0 | 0.838 | 0 | 0.794 | 0 |
| Tephrosia candida | 0.766 | 0 | 0.712 | 0 | 0.839 | 0 | 0.839 | 0 |
| Tephrosia grandiflora | 0.937 | 0 | 0.89 | 0 | 0.95 | 0 | 0.938 | 0 |
| Tephrosia purpurea | 0.756 | 0 | 0.763 | 0 | 0.82 | 0 | 0.809 | 0 |
| Tephrosia vogelii | 0.806 | 0 | 0.786 | 0 | 0.854 | 0 | 0.85 | 0 |
| Terminalia catappa | 0.806 | 0 | 0.806 | 0 | 0.859 | 0 | 0.853 | 0 |
| Thalia geniculata | 0.797 | 0 | 0.8 | 0 | 0.878 | 0 | 0.87 | 0 |
| Theobroma cacao | 0.868 | 0 | 0.865 | 0 | 0.907 | 0 | 0.907 | 0 |
| Thunbergia alata | 0.744 | 0 | 0.764 | 0 | 0.862 | 0 | 0.854 | 0 |
| Thunbergia erecta | 0.778 | 0 | 0.745 | 0 | 0.786 | 0 | 0.796 | 0 |
| Thunbergia fragrans | 0.844 | 0 | 0.816 | 0 | 0.843 | 0 | 0.834 | 0 |
| Tigridia pavonia | 0.889 | 0 | 0.839 | 0 | 0.881 | 0 | 0.872 | 0 |
| Tillandsia stricta | 0.927 | 0 | 0.915 | 0 | 0.923 | 0 | 0.92 | 0 |
| Tillandsia usneoides | 0.718 | 0 | 0.75 | 0 | 0.843 | 0 | 0.831 | 40 |
| Tithonia rotundifolia | 0.767 | 0 | 0.748 | 0 | 0.835 | 0 | 0.822 | 0 |
| Toona ciliata | 0.867 | 0 | 0.856 | 0 | 0.908 | 0 | 0.901 | 0 |
| Trachelospermum asiaticum | 0.947 | 0 | 0.942 | 0 | 0.957 | 0 | 0.958 | 40 |
| Trachelospermum jasminoides | 0.768 | 0 | 0.762 | 0 | 0.86 | 0 | 0.845 | 0 |
| Tradescantia spathacea | 0.749 | 0 | 0.749 | 0 | 0.84 | 0 | 0.831 | 0 |
| Triadica sebifera | 0.893 | 0 | 0.847 | 0 | 0.928 | 0 | 0.915 | 0 |
| Trichocentrum carthagenense | 0.881 | 0 | 0.87 | 0 | 0.902 | 0 | 0.891 | 0 |
| Trichosanthes cucumerina | 0.875 | 0 | 0.903 | 0 | 0.924 | 0 | 0.914 | 0 |
| Trichosanthes kirilowii | 0.803 | 0 | 0.801 | 0 | 0.948 | 0 | 0.957 | 0 |
| Tripsacum dactyloides | 0.777 | 0 | 0.81 | 0 | 0.894 | 0 | 0.848 | 0 |

| species | GLM | | GAM | | RF | | GBM | |
|--------------------------|----------|-------------|----------|-------------|----------|-------------|----------|-------------|
| | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 | Mean TSS | % TSS < 0.5 |
| Tulbaghia violacea | 0.888 | 0 | 0.852 | 0 | 0.906 | 0 | 0.897 | 0 |
| Ullucus tuberosus | 0.935 | 0 | 0.901 | 0 | 0.954 | 0 | 0.954 | 0 |
| Ulmus parvifolia | 0.781 | 0 | 0.718 | 0 | 0.827 | 0 | 0.814 | 0 |
| Ursinia anthemoides | 0.988 | 0 | 0.944 | 0 | 0.988 | 0 | 0.976 | 0 |
| Ursinia speciosa | 0.961 | 0 | 0.946 | 0 | 0.971 | 0 | 0.962 | 0 |
| Utricularia livida | 0.862 | 0 | 0.869 | 0 | 0.918 | 0 | 0.886 | 63.3 |
| Vallisneria americana | 0.751 | 0 | 0.838 | 0 | 0.876 | 0 | 0.871 | 0 |
| Vallisneria nana | 0.591 | 0 | 0.824 | 0 | 0.832 | 0 | 0.816 | 20 |
| Verbena stricta | 0.944 | 0 | 0.943 | 0 | 0.98 | 0 | 0.975 | 0 |
| Vernicia fordii | 0.904 | 0 | 0.877 | 0 | 0.906 | 0 | 0.896 | 0 |
| Veronica americana | 0.748 | 0 | 0.798 | 0 | 0.895 | 0 | 0.833 | 0 |
| Viburnum dilatatum | 0.927 | 0 | 0.911 | 0 | 0.965 | 0 | 0.962 | 0 |
| Viburnum plicatum | 0.918 | 0 | 0.88 | 0 | 0.951 | 0 | 0.949 | 0 |
| Viburnum sieboldii | 0.933 | 0 | 0.918 | 0 | 0.959 | 0 | 0.938 | 0 |
| Vigna caracalla | 0.797 | 0 | 0.714 | 0 | 0.857 | 0 | 0.837 | 0 |
| Viola hederacea | 0.981 | 0 | 0.976 | 0 | 0.986 | 0 | 0.973 | 0 |
| Viola sororia | 0.835 | 0 | 0.845 | 0 | 0.924 | 0 | 0.886 | 0 |
| Vitex negundo | 0.772 | 0 | 0.782 | 0 | 0.85 | 0 | 0.835 | 0 |
| Vitis coignetiae | 0.823 | 0 | 0.848 | 0 | 0.854 | 3.3 | 0.846 | 3.3 |
| Washingtonia robusta | 0.768 | 0 | 0.856 | 0 | 0.872 | 0 | 0.847 | 0 |
| Weigela floribunda | 0.841 | 0 | 0.852 | 0 | 0.953 | 0 | 0.931 | 0 |
| Westringia fruticosa | 0.97 | 0 | 0.931 | 0 | 0.947 | 0 | 0.94 | 0 |
| Wisteria floribunda | 0.922 | 0 | 0.91 | 0 | 0.919 | 0 | 0.916 | 0 |
| Xanthosoma sagittifolium | 0.839 | 0 | 0.783 | 0 | 0.872 | 0 | 0.86 | 0 |
| Ximenia americana | 0.73 | 0 | 0.745 | 0 | 0.839 | 0 | 0.824 | 0 |
| Zapoteca portoricensis | 0.818 | 0 | 0.8 | 0 | 0.861 | 0 | 0.859 | 0 |
| Zelkova serrata | 0.912 | 0 | 0.874 | 0 | 0.916 | 0 | 0.905 | 0 |
| Zingiber officinale | 0.871 | 0 | 0.688 | 0 | 0.827 | 0 | 0.803 | 0 |
| Zingiber zerumbet | 0.821 | 0 | 0.738 | 0 | 0.83 | 0 | 0.84 | 0 |
| Zinnia angustifolia | 0.727 | 0 | 0.761 | 0 | 0.876 | 0 | 0.864 | 0 |
| Zinnia elegans | 0.465 | 66.7 | 0.624 | 0 | 0.718 | 3.3 | 0.697 | 3.3 |
| Zinnia peruviana | 0.827 | 0 | 0.845 | 0 | 0.901 | 0 | 0.888 | 0 |
| Ziziphus mauritiana | 0.713 | 0 | 0.732 | 0 | 0.796 | 0 | 0.799 | 0 |
| Ziziphus spina-christi | 0.837 | 0 | 0.776 | 0 | 0.857 | 0 | 0.843 | 0 |

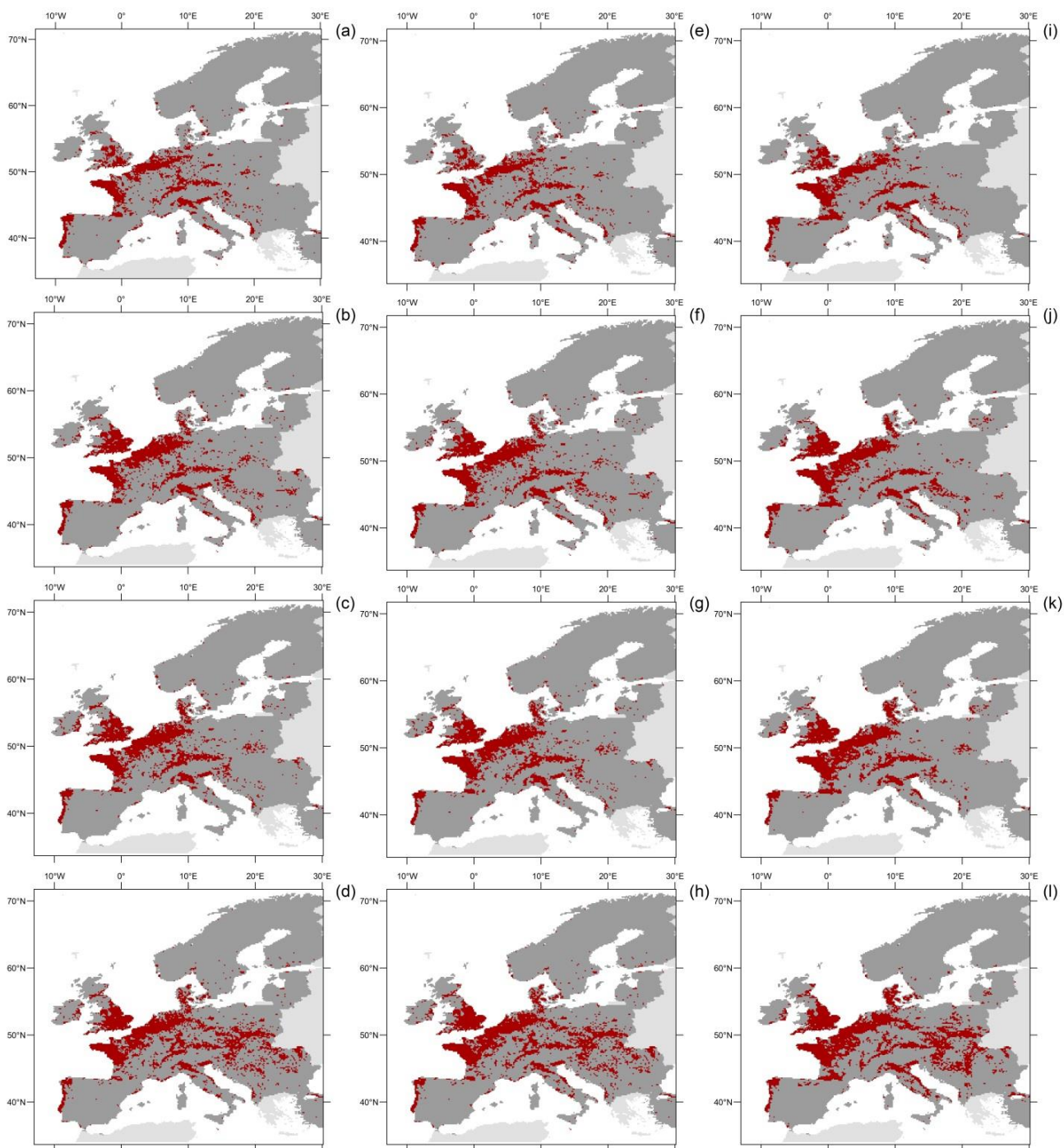
SUPPORTING INFORMATION

Climate change will increase the naturalization risk from garden plants in Europe

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Appendix S5. Naturalization risk maps calculated according to three different weighting schemes.

Naturalization risk maps calculated by combining climatic and land cover suitability for 783 ornamental species currently not naturalized in, but somewhere outside of Europe. Land cover suitability was quantified by weighting CORINE land cover types according to the estimated area available for ornamental plant cultivation according to three different weighting schemes (A: (a)-(d), B: (e)-(h), C: (i)-(l), see Appendix S3 for details). Climatic suitability is quantified by projections of species distribution models under current climate ((a), (e), (i)) and three scenarios of climate warming: mild scenario (RCP2.6: (b), (f), (j)), intermediate scenario (RCP4.5: (c), (g), (k)) and strong scenario (RCP8.5: (d), (h), (l)).



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<http://www.eea.europa.eu/publications/tech40add>

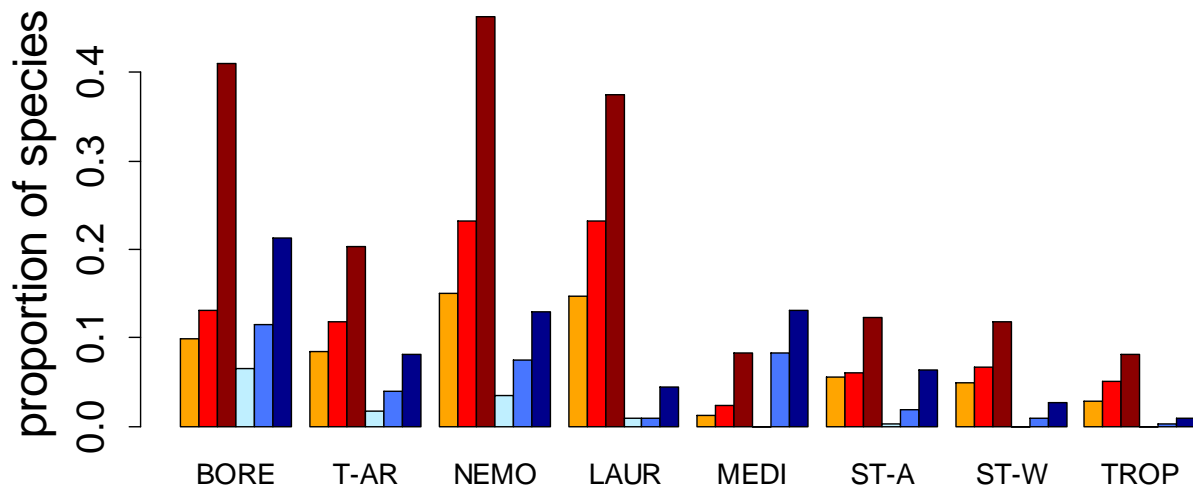
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Appendix S6. Species predicted gain or loss of area under climate change.

The proportion of species predicted to gain or lose > 1600 cells (~ 5 % of the study area) of climatically suitable area under three different climate scenarios as compared to current climatic conditions. Orange, red and dark red bars represent ‘winners’, and light blue, blue and dark blue bars represent ‘losers’ under the RCP2.6, RCP4.5, RCP8.5 scenarios, respectively. BORE: boreal, T-AR: temperate-arid, NEMO: nemoral (= temperate), LAUR: laurophyllous, MEDI: Mediterranean, ST-A: subtropical-arid, ST-W: subtropical seasonally dry, TROP: tropical.



APPENDIX MANUSCRIPT 3

(A) APPENDIX

Table S1: Overview of plant genera for which interspecific hybridization is documented, which contain garden plant species cultivated in Europe and naturalized somewhere in the world but not yet in Europe, which have at least one congener in the native or already naturalized European flora, and which are represented by > 50 occurrence records in GBIF. The taxonomically difficult genera *Rosa* and *Rubus* were excluded. FE indicates the number of congeneric species in the native or already naturalized European flora according to the Flora Europea (Tutin et al. 1964-1980). “Hybrid” indicates whether a genus contains interspecific hybrids that have naturalized somewhere (1). In addition, examples of references to articles in ISI-listed journals that report successful interspecific hybridization are given under “source”.

| genus | FE | hybrid | source |
|---------------------|-----|--------|--|
| <i>Chenopodium</i> | 27 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |
| <i>Eucalyptus</i> | 11 | 0 | e.g. (Barbour <i>et al.</i> , 2006, 2007; 2010) |
| <i>Euphorbia</i> | 106 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |
| <i>Fraxinus</i> | 5 | 1 | e.g. (Thomasset <i>et al.</i> , 2014) |
| <i>Linaria</i> | 70 | 1 | e.g. (Ward <i>et al.</i> , 2009) |
| <i>Lonicera</i> | 17 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |
| <i>Malus</i> | 6 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |
| <i>Populus</i> | 11 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |
| <i>Prunus</i> | 21 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |
| <i>Pyrus</i> | 12 | 0 | e.g. (Yamamoto <i>et al.</i> , 2002; Hardiman & Culley, 2010; Bell & Itai, 2011) |
| <i>Rhododendron</i> | 6 | 1 | e.g. (Erfmeier <i>et al.</i> , 2011) |
| <i>Senecio</i> | 67 | 1 | e.g. (Pelser <i>et al.</i> , 2012) |
| <i>Solidago</i> | 5 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |
| <i>Tamarix</i> | 14 | 1 | e.g. (Gaskin & Schaal, 2002; Gaskin & Kazmer, 2009; Lindgren <i>et al.</i> , 2010; Mayonde <i>et al.</i> , 2015) |
| <i>Ulmus</i> | 6 | 1 | e.g. (Zalapa <i>et al.</i> , 2010) |
| <i>Viola</i> | 92 | 1 | e.g. (Bleeker <i>et al.</i> , 2007) |

Table S2: List of model species (i.e. those alien garden plants cultivated in Europe and naturalized somewhere in the world but not yet in Europe, which have at least one congener in the native or already naturalized European flora, which stem from genera for which hybridization is documented) and which are represented by > 50 occurrence records in GBIF together with their life form (cf. Tab. 3) and their native range (GloNAF; van Kleunen *et al.*, 2015; <https://glonaf.org/>).

| spec | life form | native range | GBIF points |
|--------------------------------|------------------|---------------------------|--------------------|
| <i>Chenopodium quinoa</i> | annual | S-America | 96 |
| <i>Eucalyptus cinerea</i> | tree | Australia | 84 |
| <i>Eucalyptus cladocalyx</i> | tree | Australia | 88 |
| <i>Eucalyptus cornuta</i> | tree | Australia | 78 |
| <i>Eucalyptus dalrympleana</i> | tree | Australia | 277 |
| <i>Eucalyptus goniocalyx</i> | tree | Australia | 296 |
| <i>Eucalyptus leucoxylon</i> | tree | Australia | 382 |
| <i>Eucalyptus nitens</i> | tree | Australia | 73 |
| <i>Eucalyptus ovata</i> | tree | Australia | 407 |
| <i>Eucalyptus rubida</i> | tree | Australia | 405 |
| <i>Euphorbia balsamifera</i> | shrub | Africa (+ Canary Islands) | 69 |
| <i>Euphorbia leucocephala</i> | shrub | N-America, S-America | 52 |
| <i>Euphorbia mauritanica</i> | herb | Africa | 175 |
| <i>Euphorbia tithymaloides</i> | shrub | N-America, S-America | 188 |
| <i>Fraxinus nigra</i> | tree | N-America | 118 |
| <i>Fraxinus uhdei</i> | tree | N-America, S-America | 151 |
| <i>Linaria maroccana</i> | herb | Africa | 206 |
| <i>Lonicera sempervirens</i> | herb | N-America | 123 |
| <i>Malus prunifolia</i> | tree | Asia | 59 |
| <i>Populus acuminata</i> | tree | N-America | 65 |
| <i>Prunus munsoniana</i> | tree | N-America, | 57 |
| <i>Prunus pumila</i> | shrub | N-America | 152 |
| <i>Prunus salicina</i> | tree | Asia | 90 |
| <i>Pyrus calleryana</i> | tree | Asia | 88 |
| <i>Pyrus pyrifolia</i> | tree | Asia | 102 |
| <i>Rhododendron japonicum</i> | shrub | HYBRID | 58 |
| <i>Senecio radicans</i> | herb | Africa | 57 |
| <i>Senecio tamoides</i> | herb | Africa | 73 |
| <i>Solidago ptarmicoides</i> | herb | N-America | 89 |
| <i>Tamarix aphylla</i> | tree | Africa, Asia | 205 |
| <i>Tamarix chinensis</i> | tree | Asia | 216 |
| <i>Ulmus parvifolia</i> | tree | Asia | 146 |
| <i>Viola hederacea</i> | herb | Asia | 354 |
| <i>Viola sororia</i> | herb | N-America | 849 |

Table S3: Reclassification of life forms reported in the Global Naturalized Flora database (GloNAF; van Kleunen *et al.*, 2015; <https://glonaf.org/>) used for assessing correspondence between alien garden plants and potential hybridization partners in the resident flora of Europe.

| Life form | Conversion |
|---|------------|
| phanerophyte, macrophanerophyte, | tree |
| hemiphanerophyte, nanophanerophyte, shrub | shrub |
| shrub/vine | shrub/vine |
| Chamaephyte | scrub |
| herb, succulent herb | herb |
| forb/vine, herb + forb/herb, geophyte, forb/herb, | forb/herb |
| forb annual, therophyte | annual |

Table S4: Linear Mixed Effects Models (LMMs) testing either climate-driven changes in the overlap of climatically suitable ranges (by 2050-2100) of alien garden plants and their congeners in the naturalized and native flora of Europe; or climate-driven changes in the number of cells suitable either to the garden plants or their congeners. Overlap was measured by the True Skill statistic (TSS) and the number of overlapping cells. The column “model” gives the representation of the model in the statistical programming language R. The other columns document fixed effects estimates (est) with lower and upper .95 confidence intervals (lower; upper), standard error (std.error), degrees of freedom (df), t-values and p-values. Significant models (p-value < 0.05) are in bold.

| Model | lower | est | upper | std.error | df | t-value | p-value |
|------------------------------------|---------------|---------------|---------------|--------------|-----------|---------------|----------------|
| TSS | | | | | | | |
| RCP2.6/BASE ~ 1 genus | -0.043 | -0.020 | 0.004 | 0.009 | 18 | -2.488 | 0.023* |
| RCP4.5/ BASE ~ 1 genus | -0.049 | -0.026 | 0.004 | 0.011 | 18 | -2.465 | 0.024* |
| RCP8.5/ BASE ~ 1 genus | -0.096 | -0.061 | -0.025 | 0.017 | 18 | -3.622 | 0.0019* |
| <i>Cell overlap</i> | | | | | | | |
| RCP2.6/BASE ~ 1 genus | -183 | -6 | 171 | 83 | 15 | -0.071 | 0.944 |

| | | | | | | | |
|------------------------------------|--------------|--------------|-------------|------------|------------|---------------|----------------|
| RCP4.5/ BASE ~ 1 genus | -679 | -163 | 354 | 242 | 15 | -0.671 | 0.512 |
| RCP8.5/ BASE ~ 1 genus | -434 | -319 | 1073 | 354 | 15 | 0.903 | 0.381 |
| <i>Cells natives</i> | | | | | | | |
| RCP2.6/ BASE ~ 1 genus | 264 | 510 | 756 | 125 | 157 | 4.088 | 0.0001* |
| RCP4.5/ BASE ~ 1 genus | -1621 | -1041 | -460 | 294 | 157 | -3.540 | 0.0005* |
| RCP8.5/ BASE ~ 1 genus | -526 | 71 | 667 | 294 | 157 | -0.234 | 0.815 |
| <i>Cells potential invasives</i> | | | | | | | |
| RCP2.6/ BASE ~ 1 genus | -226 | 195 | 617 | 200 | 18 | 0.975 | 0.342 |
| RCP4.5/ BASE ~ 1 genus | -572 | 213 | 998 | 374 | 18 | 0.569 | 0.576 |
| RCP8.5/ BASE ~ 1 genus | -268 | 1713 | 3429 | 826 | 18 | 2.098 | 0.05* |

Table S5: Range overlap of 34 alien garden plants and their native and already naturalized European congeners under current climate and three climate change scenarios (by 2050-2100) measured as number of cells potentially suitable to both species in each pair. Numbers represent averages over all possible combinations of each of the listed garden plants with each of their possible congeneric species (see Table S6).

| species | base | RCP2.6 | RCP4.5 | RCP8.5 |
|--------------------------------|------|--------|--------|--------|
| <i>Chenopodium quinoa</i> | 4710 | 3912 | 3478 | 3622 |
| <i>Eucalyptus cinerea</i> | 731 | 1263 | 1411 | 1638 |
| <i>Eucalyptus cladocalyx</i> | 975 | 705 | 344 | 374 |
| <i>Eucalyptus cornuta</i> | 219 | 175 | 95 | 0 |
| <i>Eucalyptus dalrympleana</i> | 373 | 577 | 631 | 462 |
| <i>Eucalyptus goniocalyx</i> | 453 | 914 | 670 | 48 |
| <i>Eucalyptus leucoxylon</i> | 651 | 862 | 594 | 630 |
| <i>Eucalyptus nitens</i> | 513 | 842 | 874 | 399 |
| <i>Eucalyptus ovata</i> | 774 | 1217 | 939 | 236 |
| <i>Eucalyptus rubida</i> | 186 | 237 | 122 | 0 |
| <i>Euphorbia balsamifera</i> | 0 | 0 | 0 | 0 |
| <i>Euphorbia leucocephala</i> | 0 | 0 | 0 | 0 |
| <i>Euphorbia mauritanica</i> | 44 | 54 | 9 | 1 |
| <i>Euphorbia tithymaloides</i> | 16 | 7 | 6 | 0 |
| <i>Fraxinus nigra</i> | 32 | 102 | 86 | 766 |
| <i>Fraxinus uhdei</i> | 0 | 0 | 0 | 0 |
| <i>Linaria maroccana</i> | 9660 | 10280 | 8104 | 7947 |
| <i>Lonicera sempervirens</i> | 977 | 1458 | 1888 | 9 |
| <i>Malus prunifolia</i> | 7626 | 3765 | 4498 | 2 |
| <i>Populus acuminata</i> | 0 | 0 | 0 | 0 |
| <i>Prunus munsoniana</i> | 1 | 60 | 240 | 88 |
| <i>Prunus pumila</i> | 0 | 0 | 0 | 0 |
| <i>Prunus salicina</i> | 1609 | 1636 | 1951 | 844 |
| <i>Pyrus calleryana</i> | 2083 | 1924 | 2775 | 3170 |
| <i>Pyrus pyrifolia</i> | 15 | 24 | 24 | 0 |
| <i>Rhododendron japonicum</i> | 761 | 919 | 1262 | 1454 |
| <i>Senecio radicans</i> | 133 | 81 | 35 | 0 |
| <i>Senecio tamoides</i> | 110 | 153 | 170 | 0 |
| <i>Solidago ptarmicoides</i> | 216 | 353 | 526 | 1168 |
| <i>Tamarix aphylla</i> | 3357 | 2812 | 2758 | 434 |
| <i>Tamarix chinensis</i> | 1338 | 1106 | 897 | 690 |
| <i>Ulmus parvifolia</i> | 1963 | 2699 | 3703 | 2445 |
| <i>Viola hederacea</i> | 283 | 336 | 375 | 334 |
| <i>Viola sororia</i> | 1357 | 1106 | 1268 | 649 |

Table S6: List of the two species sets modelled: 1) 34 alien garden plants 2) 173 native or already naturalized plant species of Europe which are congeneric and share life forms with the 34 alien garden plants listed in Table S2, and which are represented in GBIF by > 50 occurrences. Further the table reflects model evaluation statistics for all modelled species. Reported is the mean TSS over all replicates and the percentage of replicates that have a TSS < 0.5, respectively, for each modelling technique.

| species | model life form | GLM | | GAM | | RF | | GBM | |
|--------------------------------|--------------------|-------|--------|-------|--------|-------|--------|-------|--------|
| | | TSS | %< 0.5 | TSS | %< 0.5 | TSS | %< 0.5 | TSS | %< 0.5 |
| <i>Chenopodium quinoa</i> | annual | 0.835 | 0.0 | 0.778 | 0.0 | 0.900 | 0.0 | 0.902 | 0.0 |
| <i>Eucalyptus cinerea</i> | tree | 0.921 | 0.0 | 0.897 | 0.0 | 0.910 | 0.0 | 0.908 | 0.0 |
| <i>Eucalyptus cladocalyx</i> | tree | 0.957 | 0.0 | 0.949 | 0.0 | 0.939 | 0.0 | 0.933 | 0.0 |
| <i>Eucalyptus cornuta</i> | tree | 0.975 | 0.0 | 0.955 | 0.0 | 0.979 | 0.0 | 0.971 | 3.3 |
| <i>Eucalyptus dalrympleana</i> | tree | 0.992 | 0.0 | 0.987 | 0.0 | 0.987 | 0.0 | 0.982 | 0.0 |
| <i>Eucalyptus goniocalyx</i> | tree | 0.977 | 0.0 | 0.986 | 0.0 | 0.978 | 0.0 | 0.975 | 0.0 |
| <i>Eucalyptus leucoxylon</i> | tree | 0.961 | 0.0 | 0.968 | 0.0 | 0.979 | 0.0 | 0.976 | 0.0 |
| <i>Eucalyptus nitens</i> | tree | 0.875 | 0.0 | 0.879 | 0.0 | 0.949 | 0.0 | 0.942 | 0.0 |
| <i>Eucalyptus ovata</i> | tree | 0.979 | 0.0 | 0.983 | 0.0 | 0.981 | 0.0 | 0.979 | 0.0 |
| <i>Eucalyptus rubida</i> | tree | 0.974 | 0.0 | 0.978 | 0.0 | 0.978 | 0.0 | 0.971 | 0.0 |
| <i>Euphorbia balsamifera</i> | shrub | 0.907 | 0.0 | 0.897 | 0.0 | 0.933 | 0.0 | 0.919 | 0.0 |
| <i>Euphorbia leucocephala</i> | shrub | 0.777 | 0.0 | 0.837 | 0.0 | 0.793 | 0.0 | 0.800 | 0.0 |
| <i>Euphorbia mauritanica</i> | herb | 0.942 | 0.0 | 0.960 | 0.0 | 0.959 | 0.0 | 0.947 | 0.0 |
| <i>Euphorbia tithymaloides</i> | shrub | 0.777 | 0.0 | 0.798 | 0.0 | 0.817 | 0.0 | 0.813 | 0.0 |
| <i>Fraxinus nigra</i> | tree | 0.947 | 0.0 | 0.950 | 0.0 | 0.964 | 0.0 | 0.951 | 0.0 |
| <i>Fraxinus uhdei</i> | tree | 0.831 | 0.0 | 0.822 | 0.0 | 0.889 | 0.0 | 0.881 | 0.0 |
| <i>Linaria maroccana</i> | herb | 0.824 | 0.0 | 0.854 | 0.0 | 0.915 | 0.0 | 0.906 | 0.0 |
| <i>Lonicera sempervirens</i> | herb | 0.924 | 0.0 | 0.933 | 0.0 | 0.967 | 0.0 | 0.961 | 0.0 |
| <i>Malus prunifolia</i> | tree | 0.909 | 0.0 | 0.766 | 0.0 | 0.892 | 0.0 | 0.850 | 0.0 |
| <i>Populus acuminata</i> | tree | 0.972 | 0.0 | 0.970 | 0.0 | 0.959 | 0.0 | 0.951 | 0.0 |
| <i>Prunus munsoniana</i> | tree | 0.938 | 0.0 | 0.842 | 0.0 | 0.964 | 0.0 | 0.970 | 0.0 |
| <i>Prunus pumila</i> | shrub | 0.910 | 0.0 | 0.936 | 0.0 | 0.936 | 0.0 | 0.926 | 0.0 |
| <i>Prunus salicina</i> | tree | 0.781 | 0.0 | 0.767 | 0.0 | 0.898 | 0.0 | 0.880 | 0.0 |
| <i>Pyrus calleryana</i> | tree | 0.802 | 0.0 | 0.870 | 0.0 | 0.868 | 0.0 | 0.850 | 0.0 |
| <i>Pyrus pyrifolia</i> | tree | 0.870 | 0.0 | 0.790 | 0.0 | 0.888 | 0.0 | 0.872 | 0.0 |
| <i>Rhododendron japonicum</i> | shrub | 0.893 | 0.0 | 0.792 | 0.0 | 0.947 | 0.0 | 0.922 | 0.0 |
| <i>Senecio radicans</i> | herb | 0.844 | 0.0 | 0.768 | 0.0 | 0.958 | 0.0 | 0.939 | 0.0 |
| <i>Senecio tamoides</i> | herb | 0.940 | 0.0 | 0.950 | 0.0 | 0.969 | 0.0 | 0.951 | 0.0 |
| <i>Solidago ptarmicoides</i> | herb | 0.861 | 0.0 | 0.845 | 0.0 | 0.867 | 0.0 | 0.859 | 0.0 |
| <i>Tamarix aphylla</i> | tree | 0.727 | 0.0 | 0.758 | 0.0 | 0.834 | 0.0 | 0.826 | 0.0 |
| <i>Tamarix chinensis</i> | tree | 0.722 | 0.0 | 0.799 | 0.0 | 0.836 | 0.0 | 0.824 | 0.0 |
| <i>Ulmus parvifolia</i> | tree | 0.827 | 0.0 | 0.732 | 0.0 | 0.815 | 0.0 | 0.817 | 0.0 |
| <i>Viola hederacea</i> | herb | 0.978 | 0.0 | 0.976 | 0.0 | 0.986 | 0.0 | 0.974 | 0.0 |
| <i>Viola sororia</i> | herb | 0.860 | 0.0 | 0.863 | 0.0 | 0.925 | 0.0 | 0.893 | 0.0 |
| <i>Chenopodium album</i> | annual | 0.881 | 0.0 | 0.907 | 0.0 | 0.935 | 0.0 | 0.909 | 0.0 |
| <i>Chenopodium capitatum</i> | annual | 0.724 | 0.0 | 0.794 | 0.0 | 0.887 | 0.0 | 0.848 | 0.0 |
| <i>Chenopodium ficifolium</i> | annual | 0.917 | 0.0 | 0.944 | 0.0 | 0.969 | 0.0 | 0.960 | 0.0 |
| <i>Chenopodium foliosum</i> | annual | 0.809 | 0.0 | 0.831 | 0.0 | 0.889 | 0.0 | 0.876 | 0.0 |
| <i>Chenopodium glaucum</i> | annual | 0.820 | 0.0 | 0.830 | 0.0 | 0.925 | 0.0 | 0.891 | 0.0 |
| <i>Chenopodium hybridum</i> | annual | 0.955 | 0.0 | 0.948 | 0.0 | 0.963 | 0.0 | 0.961 | 0.0 |
| <i>Chenopodium murale</i> | annual | 0.801 | 0.0 | 0.850 | 0.0 | 0.909 | 0.0 | 0.887 | 0.0 |
| <i>Chenopodium opulifolium</i> | annual | 0.867 | 0.0 | 0.884 | 0.0 | 0.932 | 0.0 | 0.898 | 0.0 |
| <i>Chenopodium polyspermum</i> | annual | 0.930 | 0.0 | 0.946 | 0.0 | 0.973 | 0.0 | 0.967 | 0.0 |
| <i>Chenopodium rubrum</i> | annual | 0.902 | 0.0 | 0.919 | 0.0 | 0.956 | 0.0 | 0.941 | 0.0 |
| <i>Chenopodium strictum</i> | annual | 0.796 | 0.0 | 0.842 | 0.0 | 0.925 | 0.0 | 0.878 | 0.0 |
| <i>Chenopodium sueticum</i> | annual | 0.942 | 0.0 | 0.956 | 0.0 | 0.972 | 0.0 | 0.964 | 0.0 |
| <i>Chenopodium urbicum</i> | annual | 0.914 | 0.0 | 0.914 | 0.0 | 0.956 | 0.0 | 0.929 | 0.0 |
| <i>Chenopodium vulvaria</i> | annual | 0.883 | 0.0 | 0.889 | 0.0 | 0.941 | 0.0 | 0.933 | 0.0 |

| | | | | | | | | | |
|----------------------------------|------------|-------|-----|-------|-----|-------|-----|-------|-----|
| <i>Eucalyptus botryoides</i> | tree | 0.913 | 0.0 | 0.896 | 0.0 | 0.940 | 0.0 | 0.928 | 0.0 |
| <i>Eucalyptus camaldulensis</i> | tree | 0.788 | 0.0 | 0.859 | 0.0 | 0.887 | 0.0 | 0.881 | 0.0 |
| <i>Eucalyptus globulus</i> | tree | 0.888 | 0.0 | 0.891 | 0.0 | 0.957 | 0.0 | 0.953 | 0.0 |
| <i>Eucalyptus gomphocephalus</i> | tree | 0.885 | 0.0 | 0.883 | 0.0 | 0.959 | 0.0 | 0.957 | 0.0 |
| <i>Eucalyptus resinifer</i> | tree | 0.911 | 0.0 | 0.887 | 0.0 | 0.936 | 0.0 | 0.933 | 0.0 |
| <i>Eucalyptus robusta</i> | tree | 0.888 | 0.0 | 0.859 | 0.0 | 0.898 | 0.0 | 0.895 | 0.0 |
| <i>Eucalyptus rudis</i> | tree | 0.892 | 0.0 | 0.898 | 0.0 | 0.936 | 0.0 | 0.933 | 0.0 |
| <i>Eucalyptus tereticornis</i> | tree | 0.924 | 0.0 | 0.938 | 0.0 | 0.946 | 0.0 | 0.937 | 0.0 |
| <i>Eucalyptus viminalis</i> | tree | 0.962 | 0.0 | 0.959 | 0.0 | 0.972 | 0.0 | 0.962 | 0.0 |
| <i>Euphorbia acanthothamnus</i> | scrub | 0.926 | 0.0 | 0.993 | 0.0 | 0.970 | 0.0 | 0.955 | 0.0 |
| <i>Euphorbia amygdaloides</i> | scrub | 0.958 | 0.0 | 0.965 | 0.0 | 0.979 | 0.0 | 0.969 | 0.0 |
| <i>Euphorbia angulata</i> | forb/herb | 0.943 | 0.0 | 0.973 | 0.0 | 0.969 | 0.0 | 0.966 | 0.0 |
| <i>Euphorbia biumbellata</i> | forb/herb | 0.946 | 0.0 | 0.920 | 0.0 | 0.956 | 0.0 | 0.949 | 0.0 |
| <i>Euphorbia boetica</i> | forb/herb | 0.939 | 0.0 | 0.940 | 0.0 | 0.982 | 0.0 | 0.968 | 0.0 |
| <i>Euphorbia brittingeri</i> | shrub | 0.970 | 0.0 | 0.978 | 0.0 | 0.967 | 0.0 | 0.963 | 0.0 |
| <i>Euphorbia characias</i> | forb/shrub | 0.957 | 0.0 | 0.952 | 0.0 | 0.969 | 0.0 | 0.962 | 0.0 |
| <i>Euphorbia clementei</i> | forb/herb | 0.980 | 0.0 | 0.927 | 0.0 | 0.973 | 0.0 | 0.973 | 0.0 |
| <i>Euphorbia cyparissias</i> | forb/herb | 0.912 | 0.0 | 0.932 | 0.0 | 0.962 | 0.0 | 0.953 | 0.0 |
| <i>Euphorbia dendroides</i> | shrub | 0.955 | 0.0 | 0.932 | 0.0 | 0.963 | 0.0 | 0.955 | 0.0 |
| <i>Euphorbia dracunculoides</i> | forb/herb | 0.761 | 0.0 | 0.777 | 0.0 | 0.810 | 0.0 | 0.810 | 0.0 |
| <i>Euphorbia dulcis</i> | forb/herb | 0.946 | 0.0 | 0.947 | 0.0 | 0.971 | 0.0 | 0.963 | 0.0 |
| <i>Euphorbia duvalii</i> | forb/herb | 0.992 | 0.0 | 0.974 | 0.0 | 0.993 | 0.0 | 0.974 | 0.0 |
| <i>Euphorbia epithymoides</i> | forb/herb | 0.913 | 0.0 | 0.909 | 0.0 | 0.949 | 0.0 | 0.940 | 0.0 |
| <i>Euphorbia esula</i> | forb/herb | 0.858 | 0.0 | 0.854 | 0.0 | 0.924 | 0.0 | 0.899 | 0.0 |
| <i>Euphorbia helioscopia</i> | forb/herb | 0.939 | 0.0 | 0.944 | 0.0 | 0.965 | 0.0 | 0.955 | 0.0 |
| <i>Euphorbia humifusa</i> | forb/herb | 0.859 | 0.0 | 0.868 | 0.0 | 0.879 | 0.0 | 0.881 | 0.0 |
| <i>Euphorbia hiberna</i> | shrub | 0.962 | 0.0 | 0.957 | 0.0 | 0.979 | 0.0 | 0.971 | 0.0 |
| <i>Euphorbia isatidifolia</i> | forb/herb | 0.940 | 0.0 | 0.939 | 0.0 | 0.984 | 0.0 | 0.972 | 0.0 |
| <i>Euphorbia lathyris</i> | forb/herb | 0.920 | 0.0 | 0.927 | 0.0 | 0.962 | 0.0 | 0.958 | 0.0 |
| <i>Euphorbia minuta</i> | forb/herb | 0.956 | 0.0 | 0.972 | 0.0 | 0.967 | 0.0 | 0.960 | 0.0 |
| <i>Euphorbia myrsinites</i> | forb/herb | 0.897 | 0.0 | 0.877 | 0.0 | 0.916 | 0.0 | 0.907 | 0.0 |
| <i>Euphorbia nevadensis</i> | forb/herb | 0.984 | 0.0 | 0.967 | 0.0 | 0.991 | 0.0 | 0.987 | 0.0 |
| <i>Euphorbia nicaeensis</i> | forb/herb | 0.924 | 0.0 | 0.924 | 0.0 | 0.961 | 0.0 | 0.955 | 0.0 |
| <i>Euphorbia oblongata</i> | forb/herb | 0.951 | 0.0 | 0.911 | 0.0 | 0.925 | 0.0 | 0.914 | 0.0 |
| <i>Euphorbia palustris</i> | forb/herb | 0.940 | 0.0 | 0.947 | 0.0 | 0.961 | 0.0 | 0.951 | 0.0 |
| <i>Euphorbia paralias</i> | forb/herb | 0.964 | 0.0 | 0.966 | 0.0 | 0.956 | 0.0 | 0.944 | 0.0 |
| <i>Euphorbia peplis</i> | forb/herb | 0.950 | 0.0 | 0.952 | 0.0 | 0.941 | 0.0 | 0.933 | 0.0 |
| <i>Euphorbia pithyusa</i> | scrub | 0.930 | 0.0 | 0.919 | 0.0 | 0.974 | 0.0 | 0.968 | 0.0 |
| <i>Euphorbia polygalifolia</i> | forb/herb | 0.836 | 0.0 | 0.893 | 0.0 | 0.905 | 0.0 | 0.885 | 0.0 |
| <i>Euphorbia portlandica</i> | shrub | 0.983 | 0.0 | 0.979 | 0.0 | 0.976 | 0.0 | 0.971 | 0.0 |
| <i>Euphorbia prostrata</i> | forb/herb | 0.594 | 0.0 | 0.692 | 0.0 | 0.812 | 0.0 | 0.774 | 0.0 |
| <i>Euphorbia pubescens</i> | forb/herb | 0.874 | 0.0 | 0.924 | 0.0 | 0.908 | 0.0 | 0.897 | 0.0 |
| <i>Euphorbia serrata</i> | forb/herb | 0.905 | 0.0 | 0.961 | 0.0 | 0.967 | 0.0 | 0.960 | 0.0 |
| <i>Euphorbia serrulata</i> | forb/herb | 0.781 | 0.0 | 0.833 | 0.0 | 0.901 | 0.0 | 0.888 | 0.0 |
| <i>Euphorbia spinosa</i> | scrub | 0.968 | 0.0 | 0.940 | 0.0 | 0.970 | 0.0 | 0.961 | 0.0 |
| <i>Euphorbia squamigera</i> | shrub | 0.981 | 0.0 | 0.975 | 0.0 | 0.968 | 0.0 | 0.961 | 0.0 |
| <i>Euphorbia taurinensis</i> | forb/herb | 0.860 | 0.0 | 0.854 | 0.0 | 0.902 | 0.0 | 0.897 | 0.0 |
| <i>Euphorbia terracina</i> | forb/herb | 0.952 | 0.0 | 0.955 | 0.0 | 0.966 | 0.0 | 0.956 | 0.0 |
| <i>Fraxinus angustifolia</i> | tree | 0.959 | 0.0 | 0.960 | 0.0 | 0.966 | 0.0 | 0.959 | 0.0 |
| <i>Fraxinus excelsior</i> | tree | 0.957 | 0.0 | 0.960 | 0.0 | 0.977 | 0.0 | 0.969 | 0.0 |
| <i>Fraxinus ornus</i> | tree | 0.934 | 0.0 | 0.944 | 0.0 | 0.962 | 0.0 | 0.957 | 0.0 |
| <i>Fraxinus pennsylvanica</i> | tree | 0.807 | 0.0 | 0.835 | 0.0 | 0.927 | 0.0 | 0.900 | 0.0 |
| <i>Linaria aeruginea</i> | forb/herb | 0.953 | 0.0 | 0.961 | 0.0 | 0.965 | 0.0 | 0.958 | 0.0 |
| <i>Linaria alpina</i> | forb/herb | 0.940 | 0.0 | 0.935 | 0.0 | 0.971 | 0.0 | 0.968 | 0.0 |
| <i>Linaria angustissima</i> | forb/herb | 0.987 | 0.0 | 0.993 | 0.0 | 0.972 | 0.0 | 0.960 | 0.0 |
| <i>Linaria canadensis</i> | annual | 0.781 | 0.0 | 0.778 | 0.0 | 0.880 | 0.0 | 0.864 | 0.0 |
| <i>Linaria genistifolia</i> | forb/herb | 0.931 | 0.0 | 0.925 | 0.0 | 0.906 | 0.0 | 0.888 | 0.0 |
| <i>Linaria incarnata</i> | forb/herb | 0.860 | 0.0 | 0.875 | 0.0 | 0.880 | 0.0 | 0.886 | 0.0 |
| <i>Linaria purpurea</i> | forb/herb | 0.971 | 0.0 | 0.975 | 0.0 | 0.974 | 0.0 | 0.967 | 0.0 |
| <i>Linaria repens</i> | forb/herb | 0.941 | 0.0 | 0.965 | 0.0 | 0.975 | 0.0 | 0.964 | 0.0 |
| <i>Linaria saxatilis</i> | forb/herb | 0.959 | 0.0 | 0.987 | 0.0 | 0.976 | 0.0 | 0.969 | 0.0 |
| <i>Linaria supina</i> | forb/herb | 0.911 | 0.0 | 0.924 | 0.0 | 0.964 | 0.0 | 0.947 | 0.0 |

| | | | | | | | | | |
|---------------------------------|------------|-------|-----|-------|-----|-------|-----|-------|------|
| <i>Linaria vulgaris</i> | forb/herb | 0.880 | 0.0 | 0.894 | 0.0 | 0.948 | 0.0 | 0.929 | 0.0 |
| <i>Lonicera etrusca</i> | shrub/vine | 0.949 | 0.0 | 0.955 | 0.0 | 0.965 | 0.0 | 0.957 | 0.0 |
| <i>Lonicera implexa</i> | shrub/vine | 0.973 | 0.0 | 0.977 | 0.0 | 0.981 | 0.0 | 0.968 | 0.0 |
| <i>Lonicera japonica</i> | shrub/vine | 0.833 | 0.0 | 0.860 | 0.0 | 0.934 | 0.0 | 0.915 | 0.0 |
| <i>Malus domestica</i> | tree | 0.900 | 0.0 | 0.911 | 0.0 | 0.941 | 0.0 | 0.926 | 0.0 |
| <i>Malus sylvestris</i> | tree | 0.931 | 0.0 | 0.953 | 0.0 | 0.974 | 0.0 | 0.962 | 0.0 |
| <i>Populus alba</i> | tree | 0.922 | 0.0 | 0.917 | 0.0 | 0.950 | 0.0 | 0.938 | 46.7 |
| <i>Populus candicans</i> | tree | 0.984 | 0.0 | 0.992 | 0.0 | 0.977 | 0.0 | 0.961 | 0.0 |
| <i>Populus canescens</i> | tree | 0.923 | 0.0 | 0.935 | 0.0 | 0.970 | 0.0 | 0.960 | 0.0 |
| <i>Populus deltoides</i> | tree | 0.769 | 0.0 | 0.830 | 0.0 | 0.935 | 0.0 | 0.910 | 0.0 |
| <i>Populus euphratica</i> | tree | 0.711 | 0.0 | 0.747 | 0.0 | 0.806 | 0.0 | 0.792 | 0.0 |
| <i>Populus grandidentata</i> | tree | 0.960 | 0.0 | 0.942 | 0.0 | 0.968 | 0.0 | 0.960 | 0.0 |
| <i>Populus nigra</i> | tree | 0.916 | 0.0 | 0.930 | 0.0 | 0.963 | 0.0 | 0.958 | 0.0 |
| <i>Populus simonii</i> | tree | 0.785 | 0.0 | 0.789 | 0.0 | 0.839 | 0.0 | 0.814 | 0.0 |
| <i>Populus tremula</i> | tree | 0.922 | 0.0 | 0.943 | 0.0 | 0.964 | 0.0 | 0.956 | 76.7 |
| <i>Prunus armeniaca</i> | tree | 0.735 | 0.0 | 0.728 | 0.0 | 0.823 | 0.0 | 0.807 | 0.0 |
| <i>Prunus avium</i> | tree | 0.953 | 0.0 | 0.953 | 0.0 | 0.966 | 0.0 | 0.957 | 0.0 |
| <i>Prunus cerasifera</i> | tree | 0.912 | 0.0 | 0.924 | 0.0 | 0.967 | 0.0 | 0.964 | 0.0 |
| <i>Prunus cerasus</i> | tree | 0.903 | 0.0 | 0.936 | 0.0 | 0.964 | 0.0 | 0.949 | 0.0 |
| <i>Prunus domestica</i> | tree | 0.918 | 0.0 | 0.936 | 0.0 | 0.961 | 0.0 | 0.947 | 0.0 |
| <i>Prunus dulcis</i> | tree | 0.896 | 0.0 | 0.901 | 0.0 | 0.929 | 0.0 | 0.920 | 0.0 |
| <i>Prunus fruticosa</i> | tree | 0.774 | 0.0 | 0.794 | 0.0 | 0.927 | 0.0 | 0.917 | 0.0 |
| <i>Prunus laurocerasus</i> | shrub | 0.962 | 0.0 | 0.957 | 0.0 | 0.980 | 0.0 | 0.971 | 0.0 |
| <i>Prunus lusitanica</i> | shrub | 0.940 | 0.0 | 0.956 | 0.0 | 0.976 | 0.0 | 0.972 | 0.0 |
| <i>Prunus mahaleb</i> | tree | 0.915 | 0.0 | 0.912 | 0.0 | 0.950 | 0.0 | 0.940 | 0.0 |
| <i>Prunus padus</i> | tree | 0.927 | 0.0 | 0.932 | 0.0 | 0.969 | 0.0 | 0.955 | 0.0 |
| <i>Prunus persica</i> | tree | 0.789 | 0.0 | 0.801 | 0.0 | 0.881 | 0.0 | 0.866 | 0.0 |
| <i>Prunus prostrata</i> | shrub | 0.935 | 0.0 | 0.936 | 0.0 | 0.950 | 0.0 | 0.948 | 0.0 |
| <i>Prunus serotina</i> | tree | 0.902 | 0.0 | 0.930 | 0.0 | 0.959 | 0.0 | 0.952 | 0.0 |
| <i>Prunus spinosa</i> | tree | 0.961 | 0.0 | 0.961 | 0.0 | 0.980 | 0.0 | 0.970 | 0.0 |
| <i>Prunus virginiana</i> | tree | 0.801 | 0.0 | 0.825 | 0.0 | 0.928 | 0.0 | 0.874 | 0.0 |
| <i>Pyrus bourgaeana</i> | tree | 0.979 | 0.0 | 0.978 | 0.0 | 0.982 | 0.0 | 0.972 | 0.0 |
| <i>Pyrus pyraeaster</i> | tree | 0.967 | 0.0 | 0.966 | 0.0 | 0.980 | 0.0 | 0.972 | 0.0 |
| <i>Rhododendron ferrugineum</i> | shrub | 0.978 | 0.0 | 0.964 | 0.0 | 0.983 | 0.0 | 0.972 | 0.0 |
| <i>Rhododendron hirsutum</i> | shrub | 0.973 | 0.0 | 0.976 | 0.0 | 0.964 | 0.0 | 0.958 | 0.0 |
| <i>Rhododendron lapponicum</i> | shrub | 0.754 | 0.0 | 0.764 | 0.0 | 0.887 | 0.0 | 0.870 | 0.0 |
| <i>Rhododendron luteum</i> | shrub | 0.947 | 0.0 | 0.928 | 0.0 | 0.949 | 0.0 | 0.937 | 0.0 |
| <i>Rhododendron ponticum</i> | shrub | 0.956 | 0.0 | 0.957 | 0.0 | 0.976 | 0.0 | 0.965 | 0.0 |
| <i>Senecio cacaliaster</i> | forb/herb | 0.993 | 0.0 | 0.996 | 0.0 | 0.974 | 0.0 | 0.965 | 0.0 |
| <i>Senecio carpetanus</i> | forb/herb | 0.991 | 0.0 | 0.993 | 0.0 | 0.990 | 0.0 | 0.979 | 0.0 |
| <i>Senecio doricum</i> | forb/herb | 0.941 | 0.0 | 0.949 | 0.0 | 0.954 | 0.0 | 0.945 | 0.0 |
| <i>Senecio erucifolius</i> | forb/herb | 0.967 | 0.0 | 0.962 | 0.0 | 0.987 | 0.0 | 0.982 | 0.0 |
| <i>Senecio inaequidens</i> | forb/herb | 0.969 | 0.0 | 0.971 | 0.0 | 0.980 | 0.0 | 0.973 | 0.0 |
| <i>Senecio lagascanus</i> | forb/herb | 0.966 | 0.0 | 0.942 | 0.0 | 0.977 | 0.0 | 0.972 | 0.0 |
| <i>Senecio lividus</i> | annual | 0.942 | 0.0 | 0.953 | 0.0 | 0.965 | 0.0 | 0.956 | 0.0 |
| <i>Senecio nebrodensis</i> | forb/herb | 0.927 | 0.0 | 0.924 | 0.0 | 0.929 | 0.0 | 0.911 | 0.0 |
| <i>Senecio nemorensis</i> | forb/herb | 0.899 | 0.0 | 0.907 | 0.0 | 0.932 | 0.0 | 0.925 | 0.0 |
| <i>Senecio nevadensis</i> | forb/herb | 0.954 | 0.0 | 0.985 | 0.0 | 0.968 | 0.0 | 0.955 | 0.0 |
| <i>Senecio pyrenaicus</i> | forb/herb | 0.985 | 0.0 | 0.982 | 0.0 | 0.970 | 0.0 | 0.959 | 0.0 |
| <i>Senecio smithii</i> | forb/herb | 0.991 | 0.0 | 0.945 | 0.0 | 0.988 | 0.0 | 0.995 | 0.0 |
| <i>Senecio squalidus</i> | forb/herb | 0.939 | 0.0 | 0.946 | 0.0 | 0.956 | 0.0 | 0.948 | 0.0 |
| <i>Senecio sylvaticus</i> | forb/herb | 0.937 | 0.0 | 0.952 | 0.0 | 0.960 | 0.0 | 0.948 | 0.0 |
| <i>Senecio viscosus</i> | forb/herb | 0.934 | 0.0 | 0.956 | 0.0 | 0.970 | 0.0 | 0.964 | 0.0 |
| <i>Senecio vulgaris</i> | forb/herb | 0.894 | 0.0 | 0.928 | 0.0 | 0.947 | 0.0 | 0.930 | 0.0 |
| <i>Solidago canadensis</i> | forb/herb | 0.898 | 0.0 | 0.905 | 0.0 | 0.946 | 0.0 | 0.935 | 0.0 |
| <i>Solidago gigantea</i> | forb/herb | 0.903 | 0.0 | 0.906 | 0.0 | 0.950 | 0.0 | 0.944 | 0.0 |
| <i>Solidago sempervirens</i> | forb/herb | 0.885 | 0.0 | 0.895 | 0.0 | 0.912 | 0.0 | 0.905 | 0.0 |
| <i>Solidago virgaurea</i> | forb/herb | 0.924 | 0.0 | 0.944 | 0.0 | 0.964 | 0.0 | 0.951 | 0.0 |
| <i>Tamarix africana</i> | tree | 0.942 | 0.0 | 0.941 | 0.0 | 0.958 | 0.0 | 0.950 | 0.0 |
| <i>Tamarix boveana</i> | tree | 0.991 | 0.0 | 0.992 | 0.0 | 0.990 | 0.0 | 0.990 | 0.0 |
| <i>Tamarix canariensis</i> | tree | 0.947 | 0.0 | 0.957 | 0.0 | 0.952 | 0.0 | 0.935 | 0.0 |
| <i>Tamarix gallica</i> | tree | 0.898 | 0.0 | 0.901 | 0.0 | 0.916 | 0.0 | 0.905 | 0.0 |

| | | | | | | | | | |
|------------------------------|-----------|-------|-----|-------|-----|-------|-----|-------|------|
| <i>Tamarix parviflora</i> | tree | 0.781 | 0.0 | 0.782 | 0.0 | 0.910 | 0.0 | 0.890 | 0.0 |
| <i>Tamarix ramosissima</i> | tree | 0.740 | 0.0 | 0.769 | 0.0 | 0.883 | 0.0 | 0.857 | 0.0 |
| <i>Ulmus glabra</i> | tree | 0.965 | 0.0 | 0.968 | 0.0 | 0.967 | 0.0 | 0.961 | 0.0 |
| <i>Ulmus laevis</i> | tree | 0.922 | 0.0 | 0.928 | 0.0 | 0.968 | 0.0 | 0.960 | 0.0 |
| <i>Ulmus minor</i> | tree | 0.951 | 0.0 | 0.950 | 0.0 | 0.976 | 0.0 | 0.968 | 0.0 |
| <i>Viola arvensis</i> | forb/herb | 0.923 | 0.0 | 0.924 | 0.0 | 0.957 | 0.0 | 0.947 | 0.0 |
| <i>Viola biflora</i> | forb/herb | 0.927 | 0.0 | 0.940 | 0.0 | 0.960 | 0.0 | 0.950 | 0.0 |
| <i>Viola bubanii</i> | forb/herb | 0.992 | 0.0 | 0.993 | 0.0 | 0.969 | 0.0 | 0.960 | 0.0 |
| <i>Viola calcarata</i> | forb/herb | 0.956 | 0.0 | 0.952 | 0.0 | 0.946 | 0.0 | 0.934 | 0.0 |
| <i>Viola canina</i> | forb/herb | 0.907 | 0.0 | 0.931 | 0.0 | 0.964 | 0.0 | 0.950 | 0.0 |
| <i>Viola cenisia</i> | forb/herb | 0.996 | 0.0 | 0.971 | 0.0 | 0.961 | 0.0 | 0.953 | 0.0 |
| <i>Viola collina</i> | forb/herb | 0.917 | 0.0 | 0.923 | 0.0 | 0.942 | 0.0 | 0.938 | 0.0 |
| <i>Viola cornuta</i> | forb/herb | 0.923 | 0.0 | 0.933 | 0.0 | 0.947 | 0.0 | 0.943 | 0.0 |
| <i>Viola elatior</i> | forb/herb | 0.906 | 0.0 | 0.930 | 0.0 | 0.972 | 0.0 | 0.964 | 0.0 |
| <i>Viola epipsila</i> | forb/herb | 0.892 | 0.0 | 0.958 | 0.0 | 0.973 | 0.0 | 0.962 | 0.0 |
| <i>Viola hirta</i> | forb/herb | 0.976 | 0.0 | 0.980 | 0.0 | 0.987 | 0.0 | 0.981 | 0.0 |
| <i>Viola jordanii</i> | forb/herb | 0.864 | 0.0 | 0.832 | 0.0 | 0.957 | 0.0 | 0.953 | 0.0 |
| <i>Viola kitaibeliana</i> | forb/herb | 0.939 | 0.0 | 0.947 | 0.0 | 0.964 | 0.0 | 0.950 | 0.0 |
| <i>Viola lactea</i> | forb/herb | 0.960 | 0.0 | 0.964 | 0.0 | 0.983 | 0.0 | 0.974 | 0.0 |
| <i>Viola lutea</i> | forb/herb | 0.993 | 0.0 | 0.993 | 0.0 | 0.990 | 0.0 | 0.984 | 0.0 |
| <i>Viola mirabilis</i> | forb/herb | 0.935 | 0.0 | 0.956 | 0.0 | 0.970 | 0.0 | 0.959 | 0.0 |
| <i>Viola odorata</i> | forb/herb | 0.932 | 0.0 | 0.940 | 0.0 | 0.962 | 0.0 | 0.948 | 0.0 |
| <i>Viola palustris</i> | forb/herb | 0.907 | 0.0 | 0.937 | 0.0 | 0.966 | 0.0 | 0.952 | 0.0 |
| <i>Viola parvula</i> | forb/herb | 0.974 | 0.0 | 0.947 | 0.0 | 0.937 | 0.0 | 0.928 | 0.0 |
| <i>Viola persicifolia</i> | forb/herb | 0.960 | 0.0 | 0.977 | 0.0 | 0.980 | 0.0 | 0.973 | 0.0 |
| <i>Viola pumila</i> | forb/herb | 0.939 | 0.0 | 0.959 | 0.0 | 0.987 | 0.0 | 0.979 | 0.0 |
| <i>Viola pyrenaica</i> | forb/herb | 0.978 | 0.0 | 0.981 | 0.0 | 0.971 | 0.0 | 0.965 | 0.0 |
| <i>Viola reichenbachiana</i> | forb/herb | 0.979 | 0.0 | 0.981 | 0.0 | 0.984 | 0.0 | 0.978 | 36.7 |
| <i>Viola riviniana</i> | forb/herb | 0.933 | 0.0 | 0.951 | 0.0 | 0.961 | 0.0 | 0.955 | 36.7 |
| <i>Viola rupestris</i> | forb/herb | 0.889 | 0.0 | 0.908 | 0.0 | 0.943 | 0.0 | 0.923 | 0.0 |
| <i>Viola selkirkii</i> | forb/herb | 0.862 | 0.0 | 0.871 | 0.0 | 0.947 | 0.0 | 0.935 | 0.0 |
| <i>Viola suavis</i> | forb/herb | 0.930 | 0.0 | 0.944 | 0.0 | 0.964 | 0.0 | 0.959 | 0.0 |
| <i>Viola tricolor</i> | forb/herb | 0.923 | 0.0 | 0.941 | 0.0 | 0.954 | 0.0 | 0.938 | 0.0 |
| <i>Viola uliginosa</i> | forb/herb | 0.949 | 0.0 | 0.956 | 0.0 | 0.978 | 0.0 | 0.963 | 0.0 |
| <i>Viola willkommii</i> | forb/herb | 0.984 | 0.0 | 0.977 | 0.0 | 0.974 | 0.0 | 0.964 | 0.0 |

Figure S1

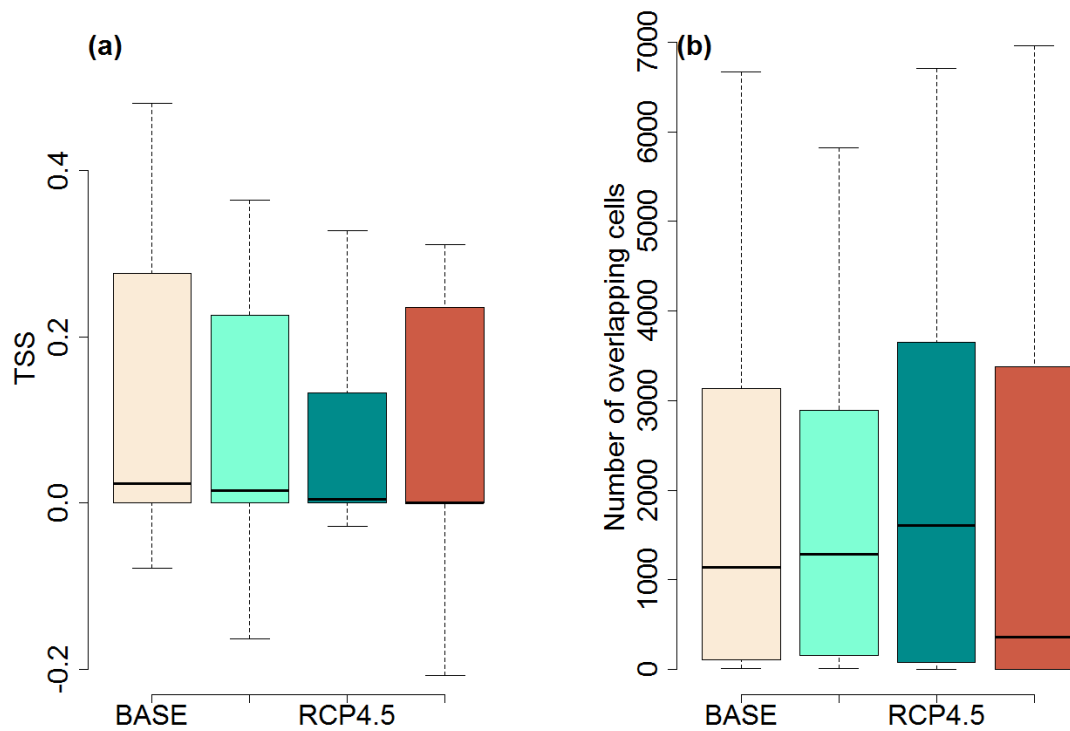


Fig. S1: Mean overlap in areas climatically suitable to 15 alien garden plants and their congeners in the native and naturalized flora of Europe. Only garden plant species native to Africa or Asia and their congeners are included. Overlap was quantified by using True Skill Statistic - TSS (A), or the number of overlapping cells (B), and calculated for current climate (BASE) and under three scenarios of climate change (RCP2.6, RCP4.5, RCP8.5) for the second half of the 21st century (2050-2100).

Figure S2

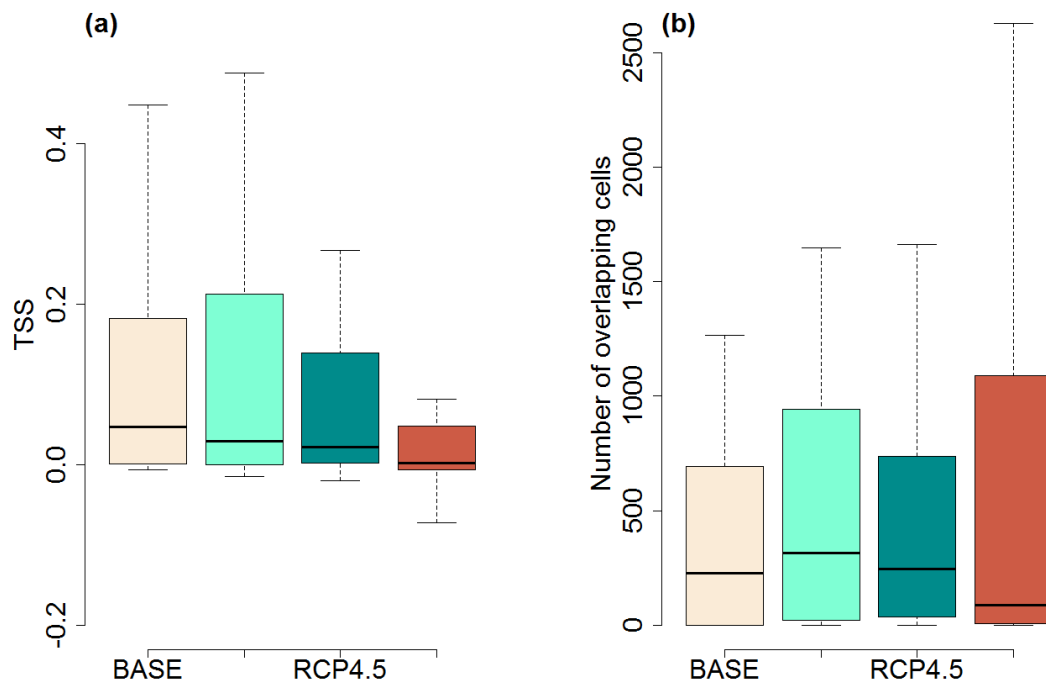


Fig. S2: Mean overlap in areas climatically suitable to 19 alien garden plants and their congeners in the native and naturalized flora of Europe. Only garden plant species not native to Africa or Asia and their congeners are included. Overlap was quantified by using True Skill Statistic - TSS (A), or the number of overlapping cells (B), and calculated for current climate (BASE) and under three scenarios of climate change (RCP2.6, RCP4.5, RCP8.5) for the second half of the 21st century (2050-2100).

Figure S3

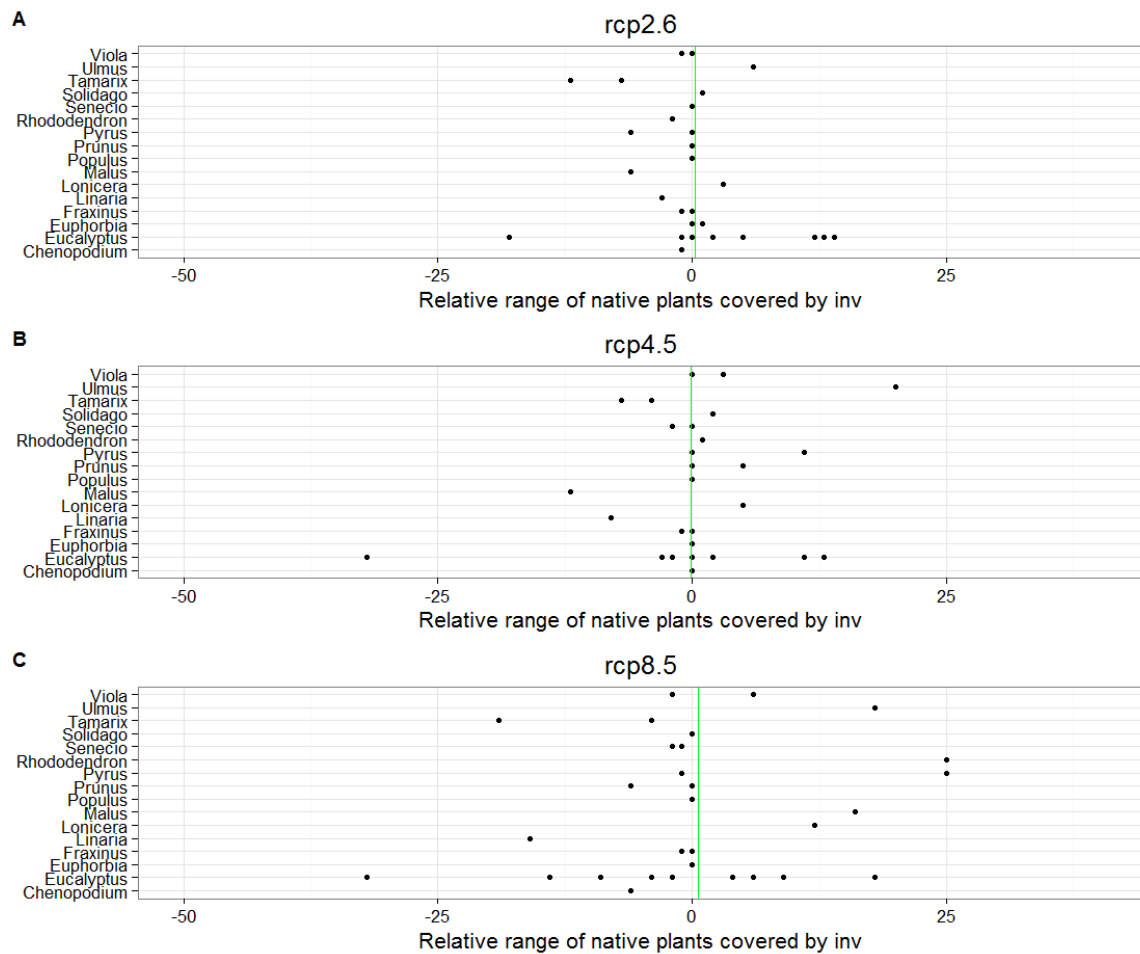


Fig. S3: Climate-driven changes in the share of the ranges of native and naturalized plant species in Europe which is also suitable to congeners among the modelled 34 alien garden plants. Share of range is measured by the ratio of 10 x 10' cells suitable to both species in a possible species pair. Each point represents the average share of range between one of the 34 garden plants and all its congeners under the respective climate scenario (some points represent more than one pair because of identical values). Values < 0 represent decreasing, values > 0 increasing, values = 0 no change in share. The three panels refer to climate change scenarios RCP2.6 (A), RCP4.5 (B) and RCP8.5 (C). The green line represents the mean over all pairs.

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APPENDIX MANUSCRIPT 4

METHODS

We simulated the possible 21st century spread of 15 ornamental plants across Europe by means of the hybrid model CATS ('Cellular Automaton-Type tool for simulating plant Spread')^{29, 30} which links simulations of demographic and dispersal processes to the output of species distribution models (SDMs) as indicators of climatic suitability. For a flow diagram summarizing the simulation approach see Supplementary Fig. S7.

STUDY REGION

The study region encompasses the member states of the European Union (except for Cyprus and Malta) plus Switzerland, Liechtenstein, Norway, Albania, Macedonia, Bosnia and Herzegovina, Serbia and Montenegro, and covers a terrestrial surface of 6.2 Mio km²

SPECIES SELECTION

Species selection focused on alien annual, perennial herbs and graminoids cultivated in Europe³¹ that naturalized elsewhere in the world, but not yet in Europe³². From the 50 species selected in a collaborative research project on potential future ornamental plant invasions (ERA-NET BiodivERsA 2012-2013) we focused on 15 species that were (i) successfully cultivated^{33, 34}, (ii) represented by at least 50 occurrence records in the Global Biodiversity Information Facility (GBIF; Supplementary Table S1) and (iii) predicted to gain climatically suitable area in Europe under at least one of three climatic scenarios tested in a previous study³⁵.

SPECIES DISTRIBUTION DATA AND CLIMATIC MAPS

We extracted data on the global distributions of the 15 selected species from GBIF (<http://www.gbif.org/>). Multiple occurrences within 10' x 10' grid cells and clearly erroneous records, e.g. in water bodies, were removed. We did not limit records to those from the native range because species are known to partly expand their realized climatic niches in the naturalization range^{36, 37}.

For characterizing current climatic conditions, we used six bioclimatic variables (data averaged for 1950-2000) provided by WorldClim³⁸ at a spatial resolution of 10': Mean Diurnal Range, Minimum Temperature of Coldest Month, Mean Temperature of Warmest Quarter, Annual Precipitation, Precipitation of Driest Month and Precipitation Seasonality. Pearson's correlations among these variables were < 0.7 throughout and thus effects of multicollinearity on model projections unproblematic³⁹.

Possible future climates in Europe were represented by three emission scenarios of the IPCC5 report⁴⁰ representing moderate (RCP 2.6), intermediate (RCP 4.5) and severe (RCP 8.5) climate change. The respective monthly temperature and precipitation time series were taken from the Cordex portal (<http://cordexesg.dmi.dk/esgf-web-fe/live>; see Supplementary Table S4) and used to recalculate 10' resolution maps of the six bioclimatic variables for future climate scenarios⁴¹. Fifteen-year running means were used to quantify temporal changes in climate over the course of the 21st century at decadal time steps (e.g. 2020 is the 15-year average for 2013-2027, etc.).

MODELLING SUITABLE AREAS

We used the BIOMOD2 framework⁴² in the programming environment R⁴³ to parameterize SDMs by correlating occurrence data from GBIF to the six bioclimatic variables. We applied the following modelling algorithms: Classification Tree Analysis (CTA), Generalized Boosted Regression Trees (GBM), Random Forest (RF), Generalized Linear Model (GLM), General Additive Model (GAM), Multiple Adaptive Regression Splines (MARS), Flexible Discriminant Analysis (FDA) and Artificial Neural Network (ANN). Pseudo-absences for these algorithms were generated by means of four approaches^{44,45}: (i) for GLM and GAM we used 10,000 randomly distributed absences; (ii) for MARS and FDA 100 randomly distributed absences; (iii) for CTA, GBM and RF as many absences as occurrences found in GBIF and selected outside of a radius of 200 km around these occurrences. In the two latter cases, absence generation and hence model calibration, was repeated ten times per species to ensure that selected absences did not bias final predictions; (iv) for ANN we used 10,000 absences selected outside of a radius of 200 km around the occurrences and repeated absence generation three times. For all models, the sum of presences was weighted equal to the sum of pseudo-absences. The predictive performance of the models was evaluated by the true skill statistic (TSS)⁴⁶ based on a three times repeated split-sampling approach in which models were calibrated with 80% of the data and evaluated on the remaining 20%. Evaluated models were then used for ensemble projections (from the techniques combined in each of the four approaches) of the area climatically suitable to each of the 15 plant species. The probabilistic output of these four ensemble models was finally aggregated to a weighted overall mean, with weights determined by the respective TSS scores. Projections were modelled for current climatic conditions (year 2010 as baseline climate) and during each decadal time step for each of the three climate change scenarios. To produce annual time series of occurrence

probabilities for the entire study area, we applied cell-wise linear interpolations of decadal projections.

SPECIES HABITAT AFFILIATIONS AND EUROPEAN HABITAT MAP

We screened online sources (Supplementary Table S3) to identify the habitat types known to be suitable to the 15 species. The suitability of these habitat types were classified using a 4-level ordinal scale (with zero meaning unsuitable and three highly suitable) for each species. We cross-tabulated these classifications to the habitat categories distinguished by CORINE land cover map (CLC, spatial resolution: 100 x 100 m²; <http://www.eea.europa.eu>; cf. Supplementary Table S7) and subsequently used CLC to create maps of the distribution of suitable habitat types for each species. As we did not account for possible land use changes, these habitat maps were kept constant across the simulation period.

SIMULATING PLANT SPREAD

The hybrid model CATS is a tool for simulating shifts in plant species' occurrence and abundance. A detailed description of the model can be found in ref. 29 and ref. 30. Briefly, the modelling framework is spatially explicit and discrete in space and time, operating at annual time steps on a two-dimensional raster (250 x 250 m² in this case), in which every cell represents an individual site. Therefore, SDM projections done at a 10' spatial resolution were resampled to a target grid consisting of 250 x 250 m² cells. Each cell was given the value of the 10' grid it was contained in. Input data for CATS are the initial distributions of the species (here: a subset of European gardens where the species are assumed to be cultivated), climatic suitability (site-specific occurrence probabilities from SDMs), habitat suitability (CLC-based habitat maps), demographic rates and the dispersal matrix (see below). By translating the probabilistic output of the SDMs into demographic rates (such as germination or juvenile survival rates), CATS re-computes the local population structure (i.e. number of seeds in the seedbank, seedlings, juveniles, adults) each year and calculates the populations' annual seed yield. Produced seeds are subsequently dispersed across the raster of sites according to dispersal kernels.

INITIAL DISTRIBUTION

We estimated the proportional area available for ornamental plant cultivation in each unit of the CLC map on a four-level scale³⁵: 0%, 0.1%, 5% or 10% (see Supplementary Table S8, Fig. S8). We further assumed that planting of ornamentals in these land cover units was spatially clustered (i.e. not every 250 x 250 m² cell of e.g. 'Sport and leisure facilities' has

ornamental plants cultivated on 0.1% of its area, but this cultivation is spatially concentrated in 0.1% of the total area of this CLC unit). We implemented this assumption by randomly subsampling from the CLC map equivalent proportions of cells from the respective CLC units for each simulation run. We henceforth refer to this subset of cells as the garden map.

To determine the potential cultivation area, we subsequently overlaid the garden map with the ensemble SDM projection, separately for each species. Because ornamental plants are often cultivated beyond the climatic conditions where they are able to sustain populations in the wild⁴⁷, climatic suitability was defined in a liberal way when determining the potential cultivation area. More precisely, we used the lowest occurrence probability predicted by ensemble SDMs for any documented presence of the species in the parameterization data (i.e. the GBIF records) as a threshold to delimit the species-specific potential cultivation area on the garden map.

Cultivation frequency levels within the potential cultivation area were derived from the proportion of European nurseries that have particular ornamental plant species on sale. The respective data were taken from ref. 47 who had analysed 13,000 ornamental plant species and their presence in 250 European nurseries. This database documents a skewed distribution with few species sold in more than 10% of the nurseries, and many available in only one or few nurseries. Based on this empirical distribution we defined six levels of cultivation frequency for our simulation design: species are cultivated in randomly selected cells totalling 0.01%, 0.1%, 0.5%, 1%, 5% and 10% of the potential cultivation area. Within the randomly selected cells, we assumed that individual plants are cultivated at an intensity of 1% of the species' climatically determined local carrying capacity.

Cultivation of species during the simulation period was implemented by accounting for changing gardening habits: each year, a certain percentage of garden owners were assumed to stop cultivation of the focal plant while others start planting it (mean cultivation time of three and ten years for annuals and perennials, respectively). In detail, a randomly selected third or tenth of the cells where the species had been cultivated in the previous year were removed from the pool of assumed planting sites and an equivalent third and tenth was randomly selected anew. The potential cultivation area was thereby adapted to the changing climate by re-projecting SDMs and repeated overlay with the garden map.

DEMOGRAPHIC RATES

Climatic dependence of local demography of each study species at each 250 x 250 m² site was modelled by linking demographic rates (germination, survival, fecundity, and clonal reproduction) to occurrence probabilities predicted by SDMs by means of sigmoidal functions.

Sigmoidal functions were confined by zero and maximum values of the respective rates and have their inflection points at occurrence probability equivalent to the threshold (found by maximizing the true skill statistic), used to translate occurrence probabilities into presence/absence values⁴⁸. At this occurrence probability, the values for demographic rates were selected such that adult population size remains constant. At lower occurrence probabilities, germination (from the seed bank or from dispersed seeds), survival and clonal reproduction were still possible, though at lower rates (according to the sigmoid functions), but seed production (fecundity) was set to zero. These populations, hence, represent remnants that decline and go extinct over time without external seed input. Decline towards extinction was accelerated when occurrence probability dropped below the value used to define suitability for cultivation of the focal species (see ‘Initial distribution’ above). At those cells, naturalized populations were assumed to decline to ½ and ¼ for the adult and the juvenile cohort, respectively, per annual time step. In addition, germination, survival and clonal reproduction were modelled as density-dependent processes to account for intraspecific competition (see ref. 30 for details on demographic modelling).

The maximum values of demographic rates were partly derived from results of own common garden experiments^{33, 34} and data gaps were complemented by information from online databases (see Supplementary Table S2). To account for uncertainty in some of the demographic rates, we assigned each species’ two sets of maximum values representing the upper and lower end of a plausible range of values (henceforth termed ‘high’ and ‘low’ values, see Supplementary Table S2), in combination with ‘high’ and ‘low’ dispersal parameter settings (see below).

CARRYING CAPACITY AND HUMAN MANAGEMENT AT RUDERAL SITES

Maximum number of ramets per cell under optimal climatic suitability (i.e. carrying capacity) was derived from the size of each species (i.e. horizontal dimensions of an individual) and the assumed maximum fraction of a cell that can be colonized by each species (Supplementary Table S9). Carrying capacity was linked to occurrence probability in the same way as demographic rates. Climatically modified carrying capacities were then further adapted according to the suitability of the local habitat type (cf. ‘Species habitat affiliations and

European habitat map' above) for the respective species. Specifically, carrying capacity was set to 0, 10, 50 and 100% of the climatically determined value in habitats of suitability class 0, 1, 2, and 3, respectively. For cultivation areas, carrying capacity of wild populations was set to zero by definition.

For land cover units defined as ruderal sites within settlements (CLC units 111, 112, 121, 122, 123, 124, 131, 132, 133, 141, 142) we assumed a certain intensity of human management (e.g. weeding, application of herbicides). Therefore, we randomly removed 80% of the populations of these sites each year. Explorative simulation runs with lower values delivered similar results, but partly higher spread rates (Supplementary Fig. S9). Presented results can hence be considered as conservative with respect to spread rates and areas simulated to be occupied.

DISPERSAL MODELLING

We modelled seed dispersal equally for all species using a compound kernel of four different vectors. Wind dispersal was implemented by means of a WALD kernel following the procedure described in ref. 30 (wind speed data of Mount Sonnblick was replaced by those of a lower lying meteorological station more representative in this context: Retz Windmühle, 15°56'35'' E, 48°45'43'' N, 320m a.s.l.). Spatio-temporal variation of wind speeds across Europe was not accounted for to keep model complexity at a feasible level and since wind dispersal contributes a minor percentage to the dispersal of herbs over longer distances, which is most relevant for species spread. We measured species properties needed for parameterizing WALD kernels (release height, seed terminal velocity) at own plant and seed material (Supplementary Table S2).

Exo- and endozoochoric dispersal kernels were derived from simulated and correlated random walks of a 'general large mammalian seed dispersal vector' following the procedure described in ref. 49. In brief, 10.000 stage-structured random walks were simulated for seeds of each of the 15 plant species. During these random walks, distances between random seed uptake points and locations where the seeds drop off from furs or are defecated again are recorded. Kernels are then derived as empirical density functions of these distances. Times until seed detachment and gut survival rates were derived from functions that relate these processes to seed mass and seed shape^{50, 51}.

Human dispersal was incorporated by using data from the OpenStreetMap⁵². The 12 classes of streets available were aggregated into highway related classes ('highways') and other paved categories ('roads'). Street map shape files were rasterized to a spatial resolution of 250 x 250

m². To simulate seed dispersal through humans along streets we randomly (uniform distribution) dispersed a proportion of seeds (i.e. human seed dispersal probability; SPD) produced in a cell containing a road into all cells containing the same street class within a given radius. For ‘highways’, maximal distance of transportation (DT) was set to 80 km, for ‘roads’ to 15 km. We evaluated the sensitivity of overall simulation results to these parameter settings in the following way (see also Supplementary Fig. S7): Comparison of simulation results (in terms of cells occupied) under constant climatic conditions for two randomly selected species (*Amaranthus tricolor*, *Helianthus debilis*) when varying SDP (0.1‰, 0.55‰, 1‰), DT for ‘highways’ (60, 80, 120 km) and DT for ‘roads’ (15, 20, 30 km). We found marginal effects of DT but more pronounced consequences of varying SDP. We hence kept DT at constant values for all simulations but varied SDP with the ‘low’ and ‘high’ dispersal setting (see below).

Exozoochoric, endozoochoric and human dispersal kernels distribute seeds over longer distances than wind. To account for the considerable uncertainty with respect to the partitioning of seed yields among the kernels we ran all simulations with a ‘high’ and a ‘low’ dispersal setting. In the ‘high’ set, between 0.5 and 1% of the seed yield was transported by each of the three kernels, while in the low set this percentage was between 0.05 and 0.1% (exact percentage defined randomly for each population and year within these ranges). Remaining seeds were dispersed by the wind kernel.

SIMULATION DESIGN AND STATISTICAL ANALYSES OF RESULTS

For each combination of species (15), climate scenario (4) and cultivation frequency (6) we ran 10 replicate simulations resulting in 3600 simulation runs per parameter set (‘high’ vs. ‘low’) and hence 7200 runs overall. We used the number of 250 x 250 m² cells occupied by a species at the end of the simulation (year 2090), except those which are part of the garden map (see ‘Initial distribution’ above), as the metric to quantify invasion success of a species in each replicate run. We compared this metric among climate scenarios and cultivation frequencies by means of linear regression models, separately for simulations under ‘high’ and ‘low’ parameter values. Prior to the analyses, the number of cells occupied in 2090 was log-transformed to achieve normality and scaled to zero mean and a standard deviation of 1, separately for the 240 replicate runs per species. This separate scaling filtered species-specific determinants of invasion success (demographic and dispersal parameters, habitat affiliations) and allowed to focus on the effects of climate change and cultivation frequency.

Linear models were run both with log-transformed or untransformed levels of cultivation frequency as predictor variable. Highly significant improvement of models after log-transformation (as evaluated by likelihood ratio tests) corroborated non-linearity of the effect of cultivation frequency on invasion success (see Fig 2 A and Supplementary Fig. S4). We partitioned the variation explained by climate change scenarios and cultivation frequency by re-calculating R²-values and Akaike Information Criterion values for models that had either one or the other of these two predictors (or their interaction terms) omitted (see Table 1 and Supplementary Table S6).

DATA AVAILABILITY

All data can be requested directly from the corresponding author.

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SUPPLEMENTARY FIGURES

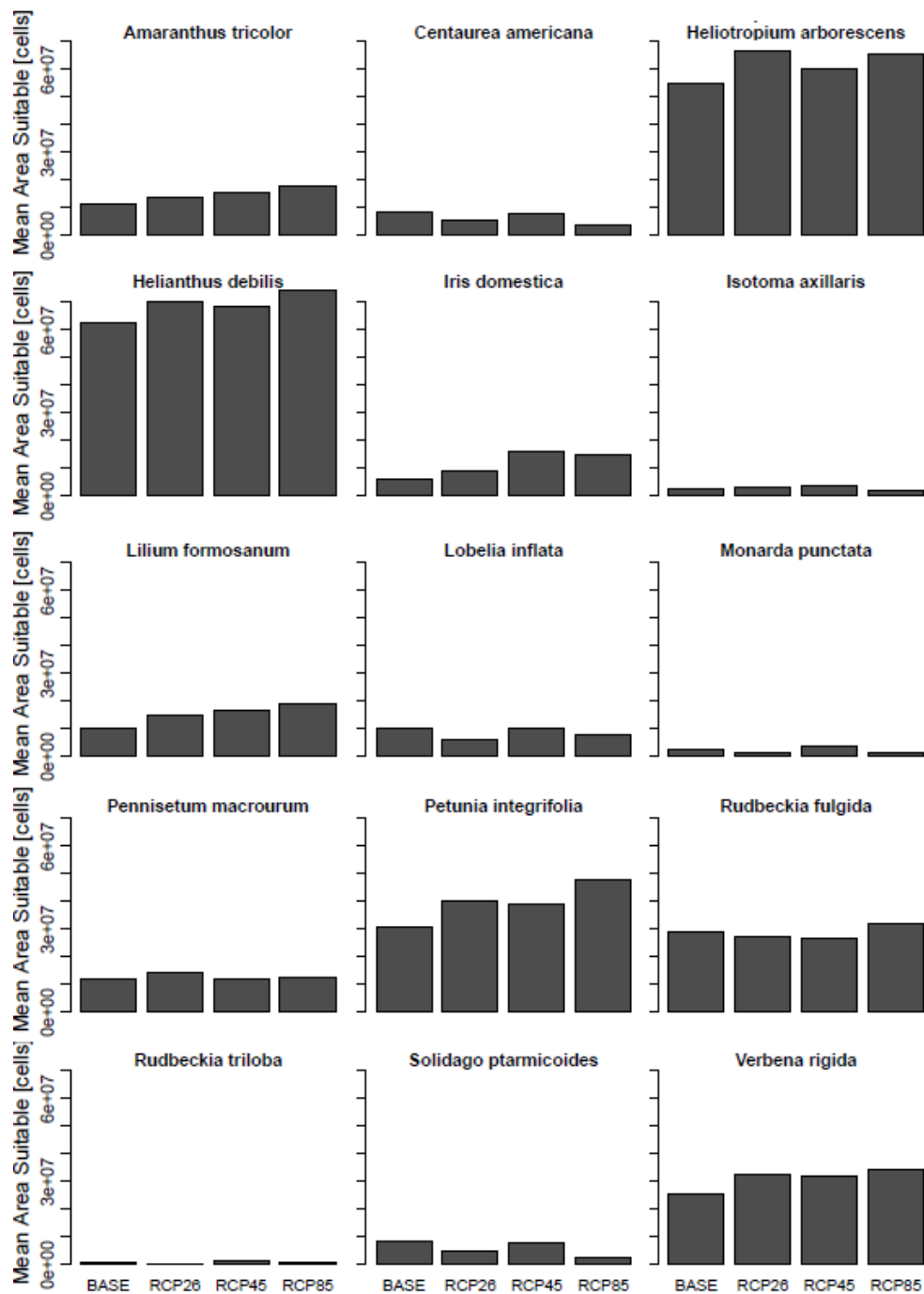


Fig. S1: Mean area climatically suitable to 15 alien ornamental plants in Europe during the 21st century under four climate change scenarios (BASE, RCP2.6, RCP4.5 and RCP8.5). The 'BASE' scenario represents a stable current climate, based on data from WorldClim¹⁰. Suitable areas are means of projections from species distribution models for the period 2020 to 2090.

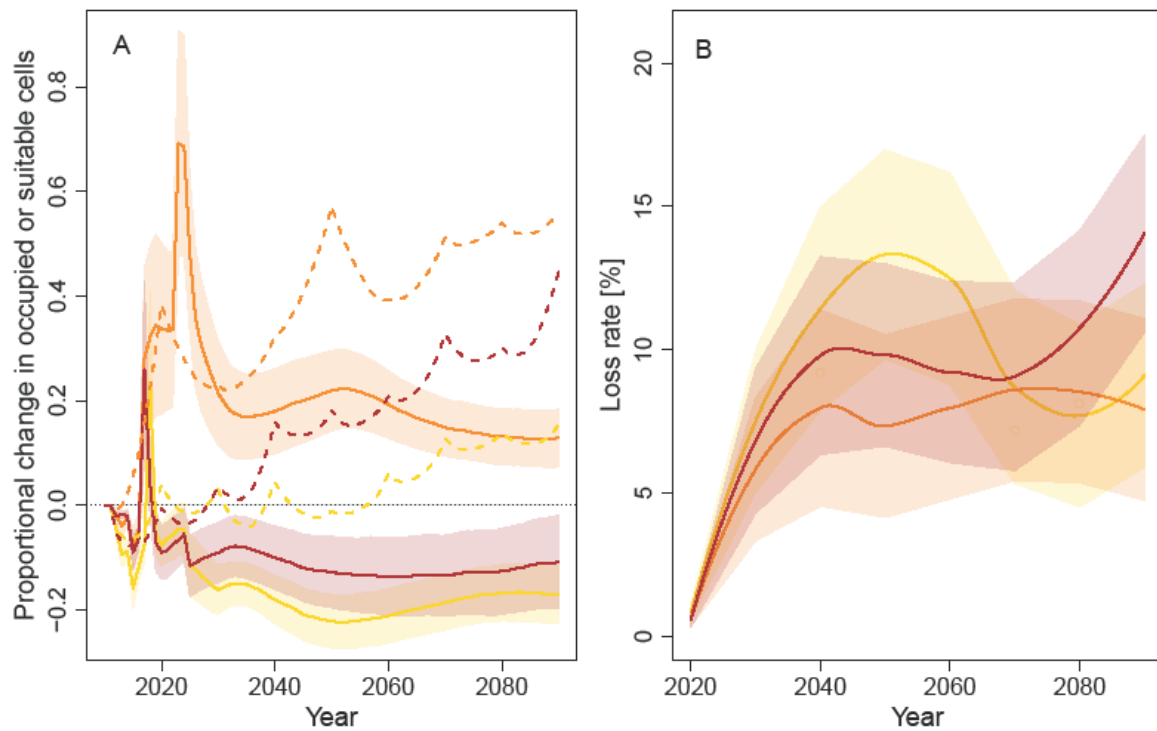


Fig. S2: Simulated future development of surface area occupied by 15 alien ornamental plants under climate change in Europe. Colours indicate different climate change scenarios: ‘yellow’ – moderate RCP2.6, ‘orange’ – intermediate RCP4.5, ‘red’ – severe RCP8.5; lines represent averages over all 15 species and shaded areas indicate standard errors. A: The proportional change in the number of cells occupied (solid lines) and climatically suitable (dashed lines). B: Percentage loss of occupied cells between two consecutive decades. The presented results are for simulations with cultivation frequency of 1 % and dispersal parameters set to ‘low’.

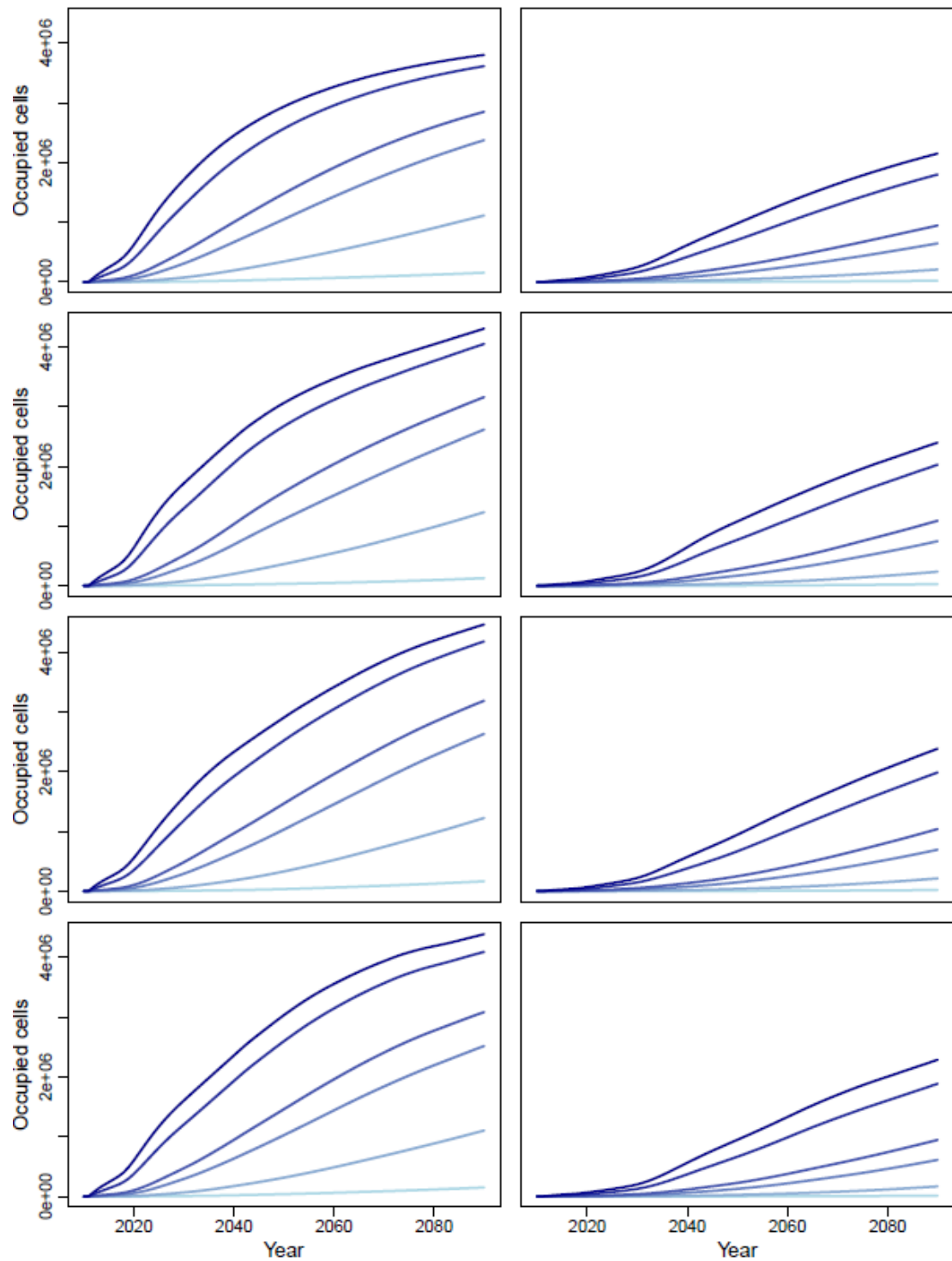


Fig. S3: Mean number of cells modelled to be occupied by 15 alien ornamental plants in Europe during the 21st century. Results are for ‘high’ (left column) and ‘low’ dispersal parameter values (right column). Rows represent the results for the different climate change scenarios, including a stable current climate scenario (top to bottom = BASE, RCP2.6, RCP4.5, RCP8.5). The lines refer to different cultivation frequencies (light blue to dark blue = 0.01 % to 10%).

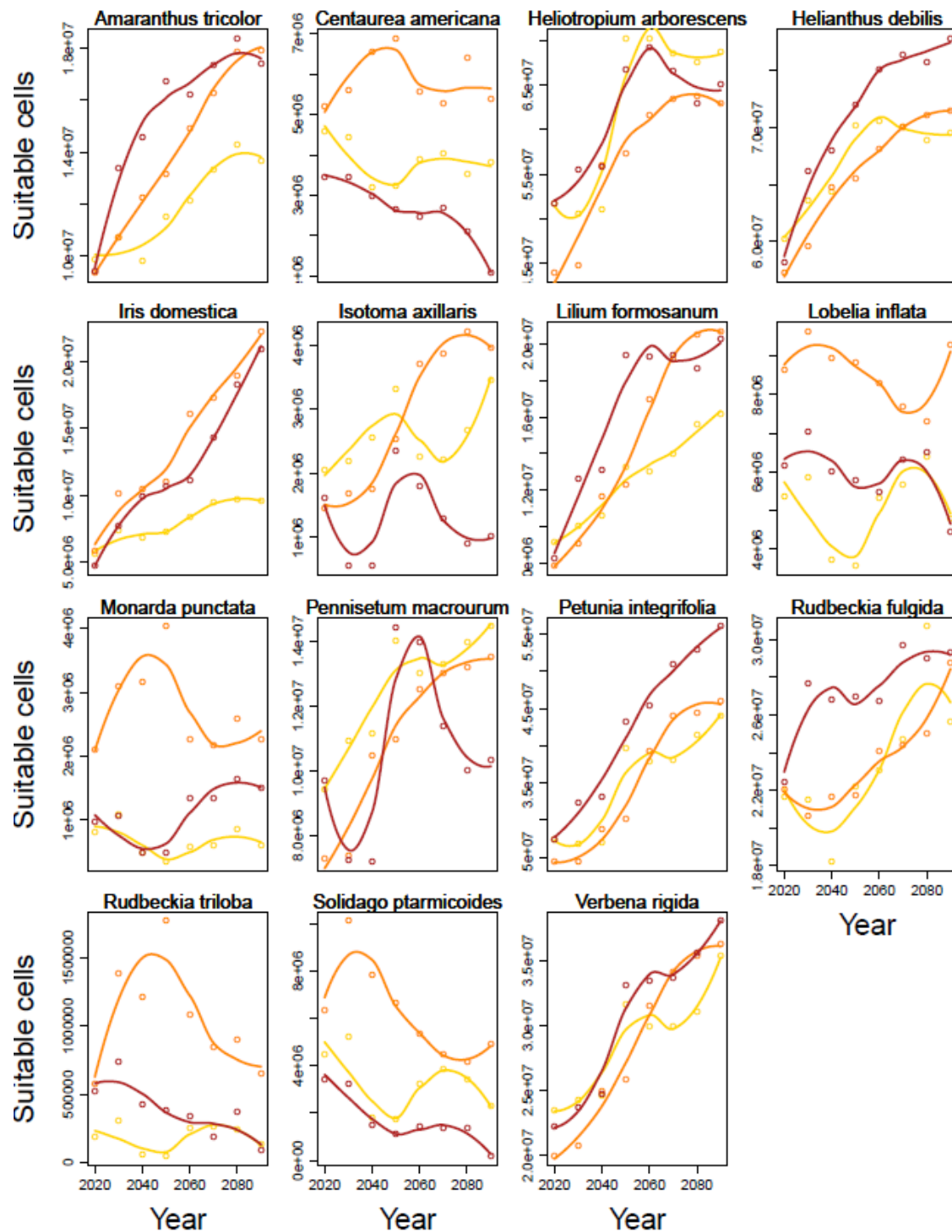


Fig.S4: Number of cells suitable to 15 alien ornamental plants in Europe at both the given and the consecutive decade under three climate change scenarios during the 21st century ('yellow' – RCP2.6, 'orange' – RCP4.5, 'red' – RCP8.5). A decreasing trend indicates that suitable areas shrink and / or shift.

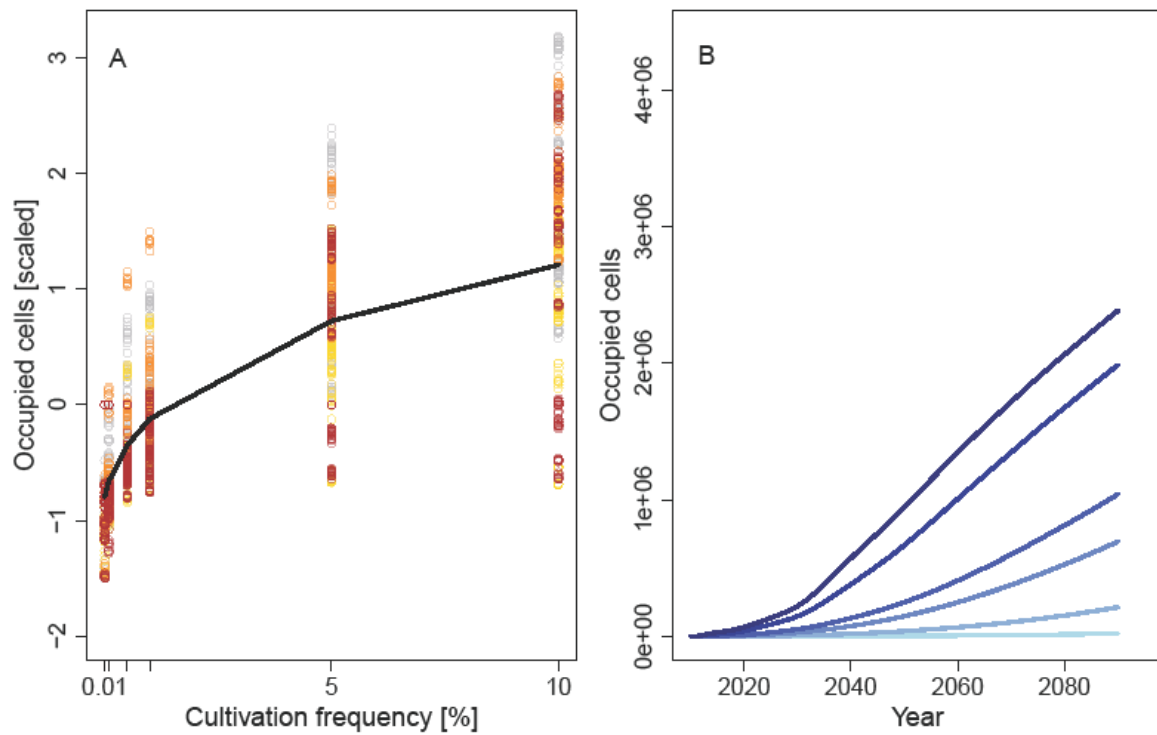


Fig. S5: Effect of cultivation frequency on the simulated spread of 15 alien ornamental plants in Europe A: The average area occupied by the species at the end of the simulation period (year 2090), measured as number of 250 x 250 m² cells. Cell numbers have been scaled to a mean of 0 and a standard deviation of 1, separately for each species. Circles represent results of individual simulation runs, with colours indicating climate change scenarios ('grey' – Baseline, 'yellow' – moderate RCP2.6, 'orange' – intermediate RCP4.5, 'red' – severe RCP8.5). B: The average number of cells occupied by the 15 species over the simulation period. Colours indicate different cultivation frequencies from 0.01 % (light blue) to 10% (dark blue). The results represent simulations under the RCP 4.5 scenario and dispersal parameters set to 'low'.

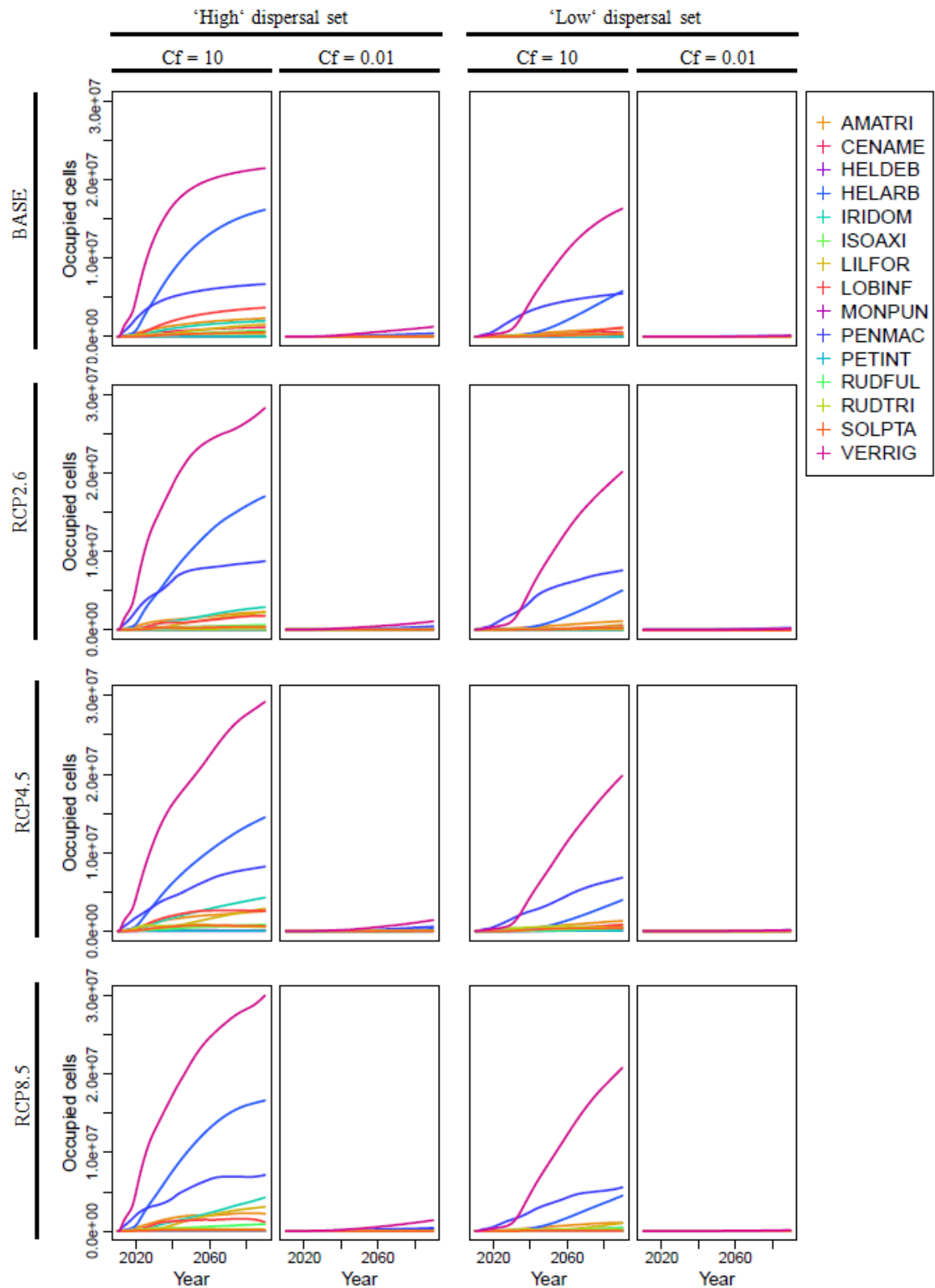


Fig. S6: Number of cells modelled to be occupied by 15 alien ornamental plants in Europe during the 21st century (2010-2090). Results are for 'high' and 'low' dispersal set and for two different levels of cultivation frequency [%] ('Cf'; Rows represent the results for the different

climate change scenarios, including a stable current climate (top to bottom = BASE, RCP2.6, RCP4.5, RCP8.5). The lines refer to the different species. For abbreviations of species names see Supplementary Table S1.

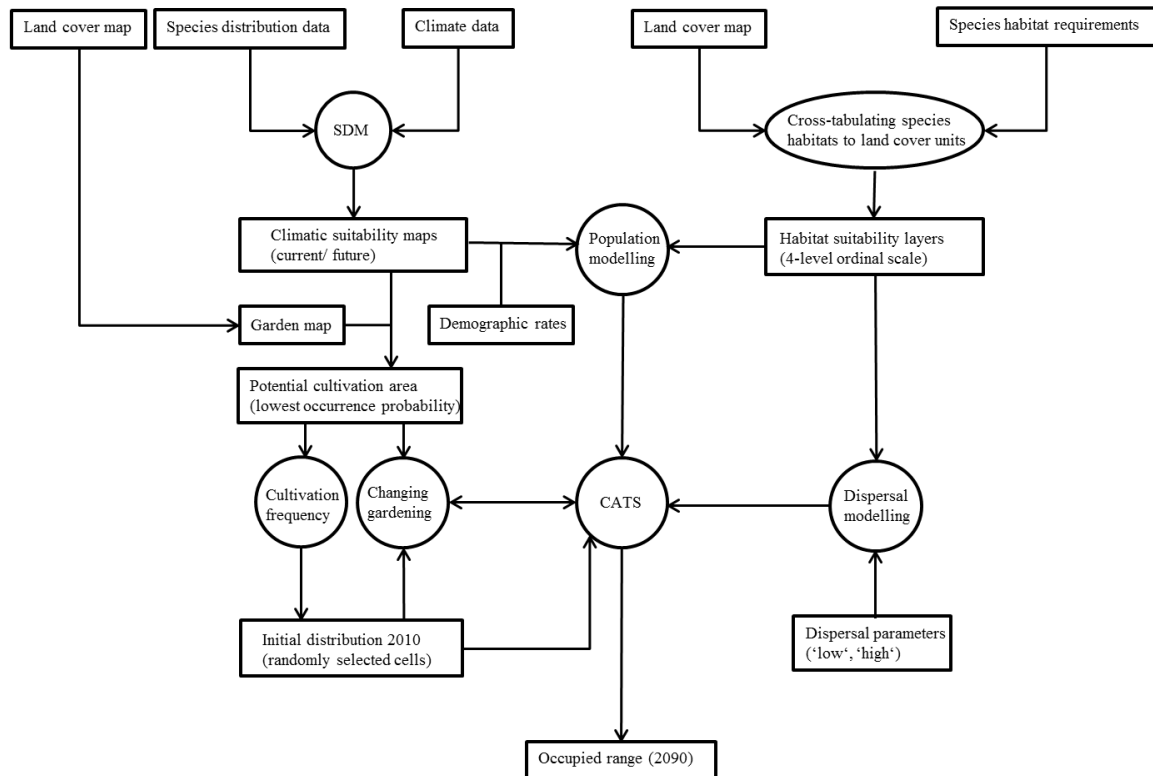


Fig. S7: Modelling steps and process flow.

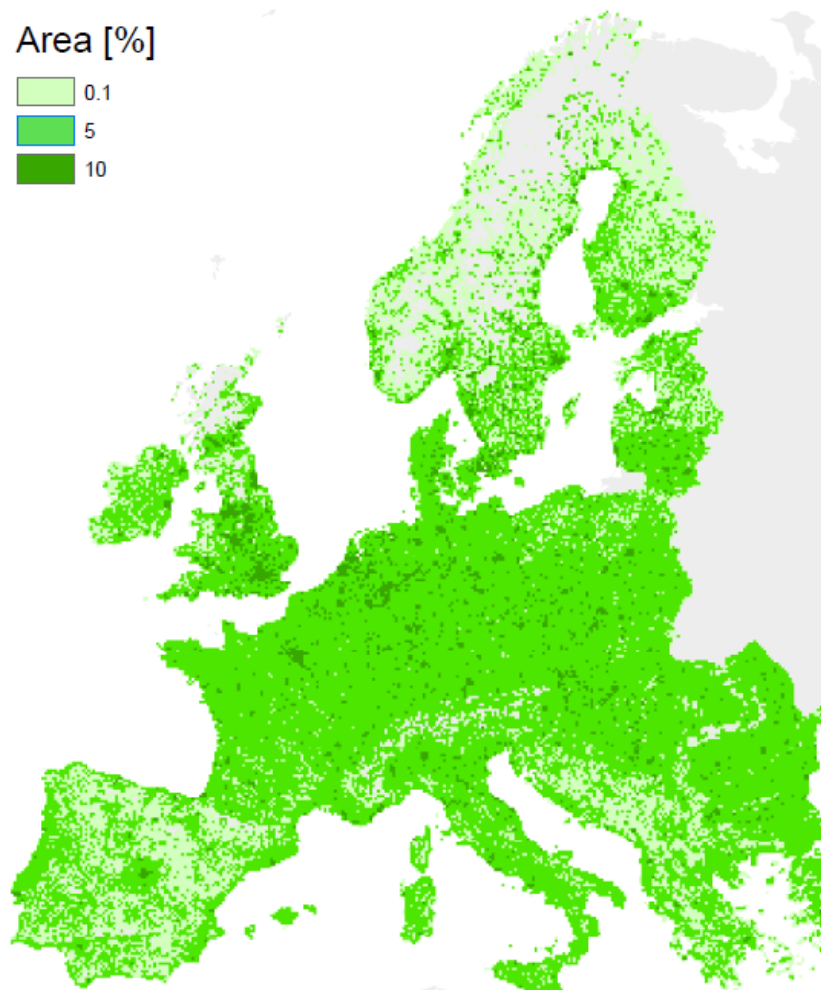
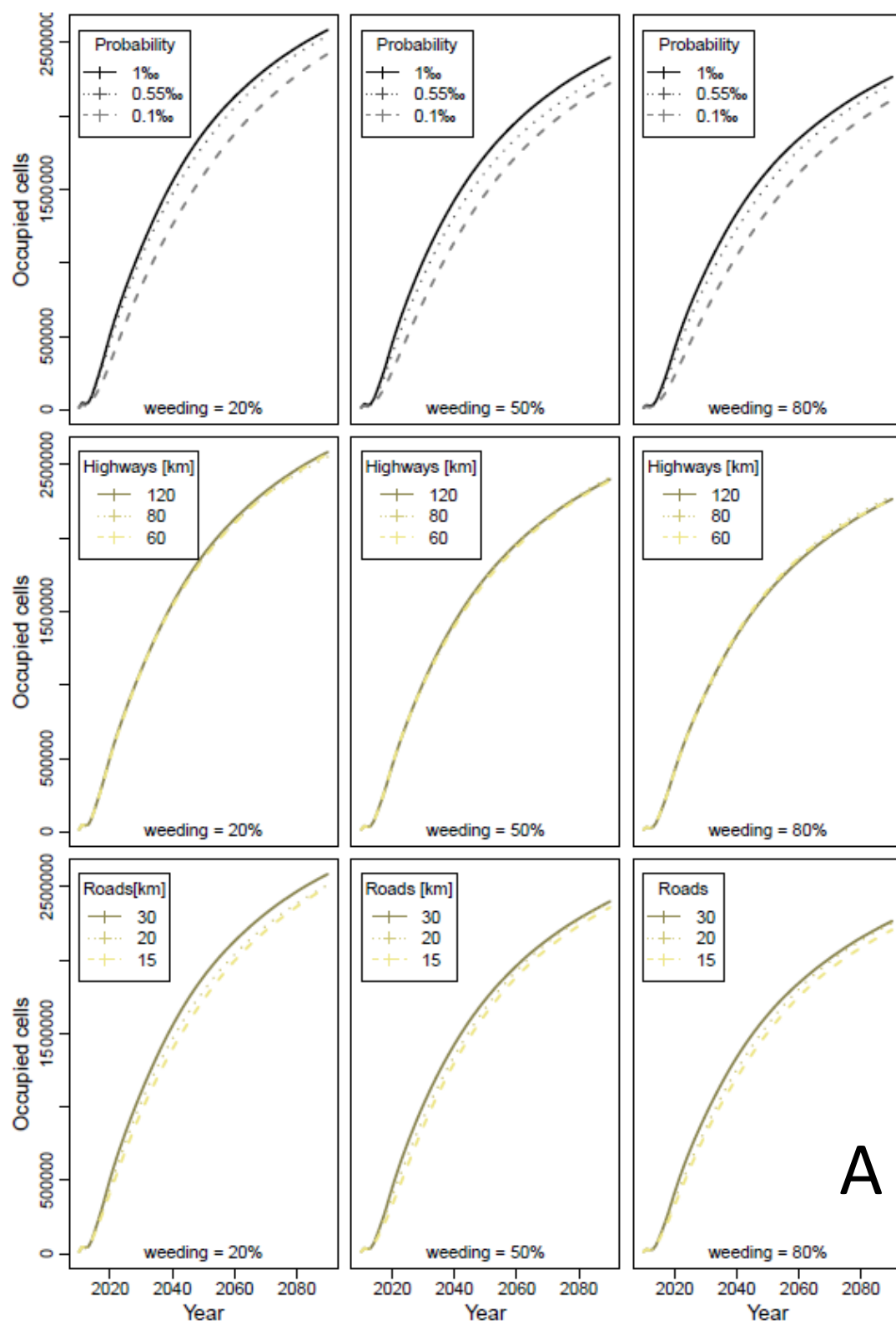


Fig. S8: Overall pool of gardens and green space area available for cultivating the model species as derived from the CORINE land cover map (<http://www.eea.europa.eu>) on a spatial resolution of 250 x 250 m². CORINE units were re-classified according to their estimated proportional area (%) available for ornamental plant cultivation⁹ (0.1, 5 or 10; see Supplementary Table S8). The graphic account was aggregated to a spatial resolution of 10 x 10 km² to enhance visibility.



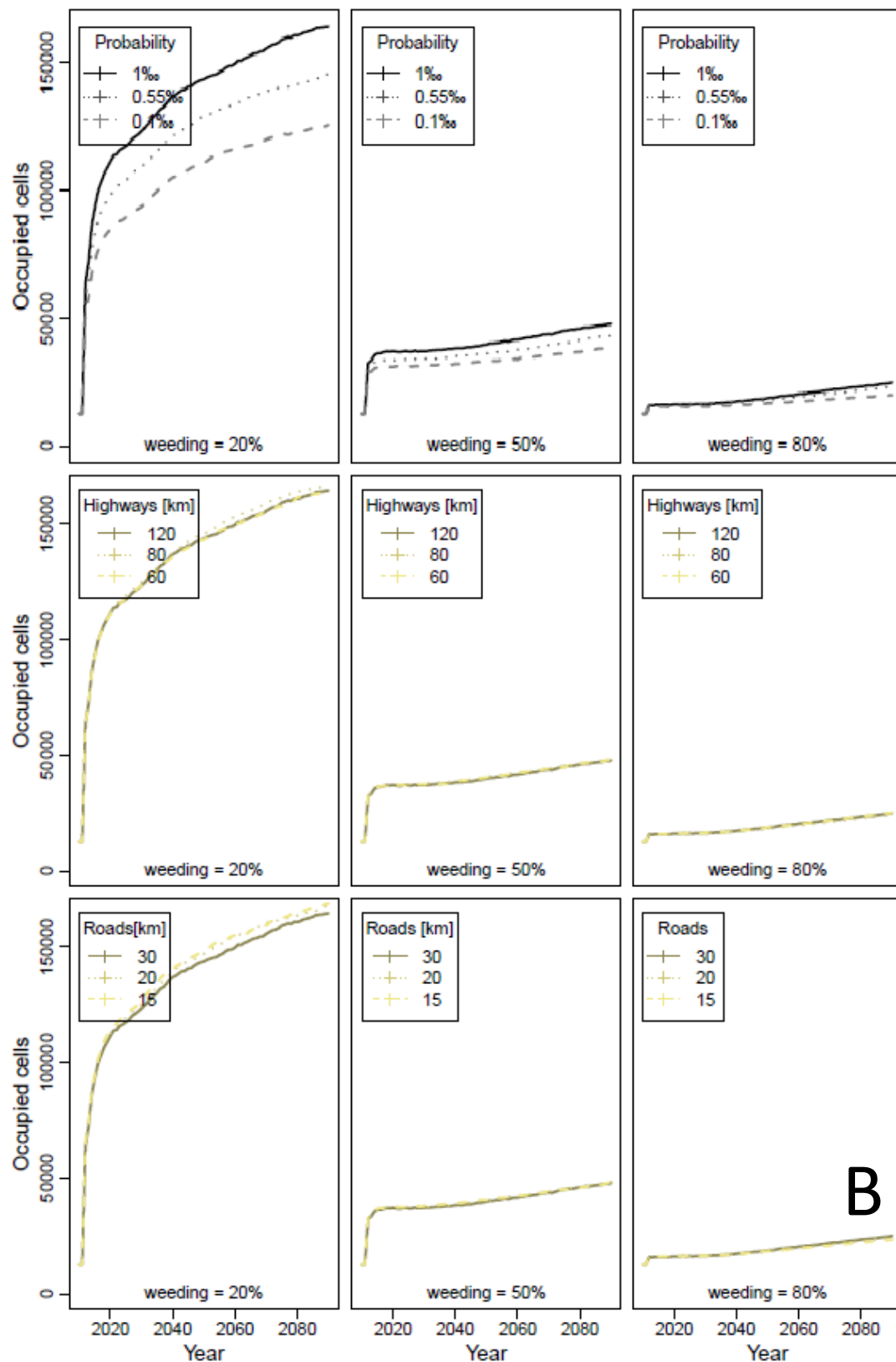


Fig. S9: Sensitivity of simulation results to different settings of human long-distance dispersal. We compared simulation results (in terms of cells occupied) under constant climatic conditions for two randomly selected species (*Amaranthus tricolor* - A, *Helianthus debilis* -

B) when varying human seed dispersal probability (0.1‰, 0.55‰, 1‰, upper panels), distance of transportation on highway related streets ('Highways'; 60km, 80km, 120km, central panels), distance of transportation on other paved streets ('Roads'; 15km, 20km, 30km, lower panels) and human management of ruderal areas ('weeding'; 20%, 50%, 80%).

SUPPLEMENTARY TABLES

Table S1: List of the 15 alien ornamental plant species modelled, together with the number of global occurrence records from the Global Biodiversity Information Facility (GBIF; www.gbif.org).

| Family | Species | Abbreviation | GBIF records |
|---------------|---------------------------------|--------------|--------------|
| Amaranthaceae | <i>Amaranthus tricolor</i> | AMATRI | 50 |
| Asteraceae | <i>Centaurea americana</i> | CENAME | 79 |
| Asteraceae | <i>Helianthus debilis</i> | HELDEB | 89 |
| Boraginaceae | <i>Heliotropium arborescens</i> | HELARB | 68 |
| Iridaceae | <i>Iris domestica</i> | IRIDOM | 117 |
| Campanulaceae | <i>Isotoma axillaris</i> | ISOAXI | 265 |
| Liliaceae | <i>Lilium formosanum</i> | LILFOR | 161 |
| Campanulaceae | <i>Lobelia inflata</i> | LOBINF | 220 |
| Lamiaceae | <i>Monarda punctata</i> | MONPUN | 165 |
| Poaceae | <i>Pennisetum macrourum</i> | PENMAC | 252 |
| Solanaceae | <i>Petunia integrifolia</i> | PETINT | 80 |
| Asteraceae | <i>Rudbeckia fulgida</i> | RUDFUL | 94 |
| Asteraceae | <i>Rudbeckia triloba</i> | RUDTRI | 179 |
| Asteraceae | <i>Solidago ptarmicoides</i> | SOLPTA | 89 |
| Verbenaceae | <i>Verbena rigida</i> | VERRIG | 512 |

Table S2: Demographic parameter values of the 15 alien ornamental species modelled. Data were derived from field experiments^{1, 2} (later on referred to as FE), and complemented by information from online databases (Encyclopaedia of Life www.eol.org; www.efloras.org; Global Invasive Species Database GISD, www.iucngisd.org).

| Species name | Mat | Age | SP | CC | CLG | FF | JS | SY | Germ | h | vt | H0 | GS | DR |
|---------------------------------|-----|-----|----|---------|---------|------|-----------|-------|--------|------|------|------|------|-------|
| <i>Amaranthus tricolor</i> | 0 | 1 | 10 | 62500 | 1.0 | 0.72 | 0.33/0.17 | 4000 | 0.2933 | 0.30 | 2.73 | 0.51 | 0.02 | 44.78 |
| <i>Centaurea americana</i> | 0 | 1 | 1 | 125000 | 1.0 | 0.72 | 0.33/0.17 | 1200 | 0.38 | 0.30 | 3.99 | 1.02 | 0.00 | 12.53 |
| <i>Helianthus debilis</i> | 1-3 | 10 | 1 | 312500 | 1.0 | 0.72 | 0.33/0.17 | 150 | 0.08 | 0.30 | 3.91 | 1.33 | 0.01 | 19.13 |
| <i>Heliotropium arborescens</i> | 1-3 | 10 | 10 | 125000 | 1.0 | 0.72 | 0.33/0.17 | 1636 | 0.06 | 0.30 | 2.34 | 0.33 | 0.02 | 43.17 |
| <i>Iris domestica</i> | 1-3 | 50 | 1 | 62500 | 6.0/3.0 | 0.72 | 0.33/0.17 | 79 | 0.1066 | 0.30 | 4.16 | 0.48 | 0.00 | 10.09 |
| <i>Isotoma axillaris</i> | 2-3 | 10 | 10 | 1250000 | 1.0 | 0.72 | 0.58/0.41 | 600 | 0.01 | 0.30 | 1.24 | 0.40 | 0.08 | 66.56 |
| <i>Lilium formosanum</i> | 2-3 | 50 | 1 | 625000 | 2.0 | 0.72 | 0.58/0.41 | 160 | 0.01 | 0.30 | 0.90 | 1.30 | 0.01 | 15.91 |
| <i>Lobelia inflata</i> | 1-2 | 50 | 10 | 62500 | 2.0 | 0.72 | 0.33/0.17 | 236 | 0.1933 | 0.30 | 1.41 | 0.57 | 0.03 | 51.66 |
| <i>Monarda punctata</i> | 1-2 | 50 | 10 | 62500 | 2.0 | 0.72 | 0.33/0.17 | 1328 | 0.26 | 0.30 | 2.47 | 1.75 | 0.03 | 51.41 |
| <i>Nemophila maculata</i> | 0 | 1 | 1 | 625000 | 1.0 | 0.72 | 0.33/0.17 | 200 | 0.02 | 0.30 | 3.71 | 0.30 | 0.01 | 20.31 |
| <i>Pennisetum macrourum</i> | 1-2 | 50 | 5 | 2500000 | 6.0/3.0 | 0.72 | 0.33/0.17 | 1243 | 0.68 | 0.30 | 1.88 | 1.49 | 0.01 | 44.91 |
| <i>Petunia integrifolia</i> | 1-2 | 10 | 10 | 1250000 | 1.0 | 0.72 | 0.33/0.17 | 40303 | 0.3333 | 0.30 | 1.87 | 0.88 | 0.09 | 69.07 |
| <i>Rudbeckia fulgida</i> | 1-2 | 10 | 1 | 93750 | 2.0 | 0.72 | 0.33/0.17 | 1544 | 0.3267 | 0.30 | 2.26 | 0.35 | 0.02 | 39.33 |
| <i>Rudbeckia triloba</i> | 1-2 | 10 | 5 | 93750 | 2.0 | 0.72 | 0.33/0.17 | 10400 | 0.34 | 0.30 | 2.43 | 1.03 | 0.02 | 44.54 |
| <i>Solidago ptarmicoides</i> | 2-3 | 50 | 1 | 93750 | 2.0 | 0.72 | 0.58/0.41 | 7000 | 0.01 | 0.30 | 2.51 | 0.30 | 0.03 | 47.45 |
| <i>Verbena rigida</i> | 1-3 | 50 | 10 | 625000 | 2.0 | 0.72 | 0.33/0.17 | 198 | 0.3267 | 0.30 | 2.89 | 0.43 | 0.02 | 40.68 |

Mat –age at maturity (minimum-maximum in years). Used as a proxy of time between germination and first seed production. A value of 0 indicates annual species. Values are from FE.

Age – maximum age of genets; four classes (1 annual, 2 biennial, 10 perennial, 50 long-lived perennial).

SP – persistence of seeds in the soil seed bank; three classes (1 transient, 5 short-term persistent, 10 long-term persistent); calculated based on seed mass and seed shape (both own measurements) according to the formula in ref. 3.

CC – carrying capacity as number of individuals (annuals) or shoots (others) within a 250 x 250 m² cell. We estimated maximum percent cover values of each species under optimal conditions based on appearance, habit and coverage values of similar plants in the native flora (see Supplementary Table S10). The number of individuals or shoots was then calculated by dividing the respective percentage of the overall cell area (250 x 250 m²) by the estimated area occupied by one individual or shoot. Resulting numbers were multiplied by the fraction of the cell assumed to be actually colonisable for the species in a typical cell of the respective habitat type (e.g. for ruderal habitats CC is divided by 10, as we assumed that 10% of ruderal areas are colonisable for plants, see Supplementary Table S7).

CLG – clonal propagation rates, i.e., number of offspring shoots per parent shoot plus 1 (representing the parental shoot). Value “1” indicates no clonal growth. Data were taken from FE and online databases. As this parameter was not well captured in FE we used a ‘high’ and a ‘low’ value in combination with the respective settings for the dispersal parameters.

FF – flowering frequency; we used a fixed value of 72% (mean value of available data in FE) for all species as FE duration was too short to assess mean flowering frequency of perennials and ornamentals are selected for high flowering frequency.

JS – juveniles surviving; % individuals surviving from one year to the next. As this parameter was not well captured in FE we used a ‘high’ and a ‘low’ value in combination with the respective settings for the dispersal parameters. For species which are able to reach adult stage after one year (or earlier) annual juvenile-survival rates were estimated to be 33%, 17% for ‘high’ or ‘low’ parameter set, respectively. For species that stay juvenile for at least two years, on the other hand, annual juvenile-survival rates were set to 58%, 41% for ‘high’ or ‘low’ parameter set, respectively, resulting in a two- year survival rate of approximately 33% and 17%.

SY – seed yield; number of seeds produced per flowering individual or shoot, calculated as number of seeds per flower multiplied by the number of flowers per individual or shoot; data were derived from FE; missing data were interpolated from average values of congeneric species⁴.

Germ – germination rates as mean percentage of seeds germinating and surviving the first five months; data from FE.

h – height of the vegetation surrounding a fruiting plant (in cm), a parameter used in fitting WALD dispersal models⁵. We set this value to a constant (30 cm) because spatially explicit information on vegetation height (herbaceous layer) of invaded communities was not available.

vt – Terminal velocity of seeds (in m/s), measured with a terminal velocity-meter⁶. Values of 10 different seeds per species were averaged.

H0 – seed release height (in m); derived from FE.

GS – Probability to survive the gut passage of a general large mammalian dispersal vector as calculated from mean seed mass of 20-50 seeds per species by using the regression equation in ref. 7.

DR – hourly detachment rate of seeds from sheep and cattle fur (in %) as calculated from seed mass and surface structure based on regression equations in⁸. Seed mass was measured for 20-50 seeds per species.

Table S3: Suitable habitats of the species studied and the source of information. All online sources were accessed between 03.08.2017 and 20.06.2016.

| species | suitable habitats | data sources |
|---------------------------------|--|---|
| <i>Amaranthus tricolor</i> | ruderal areas, agricultural areas | http://eol.org/pages/597119/data |
| <i>Centaurea americana</i> | grasslands, shrubs, forests, ruderal areas, rock and scree | http://eol.org/pages/467766/details |
| <i>Helianthus debilis</i> | dunes | http://eol.org/pages/468125/data ; http://www.floridata.com/ref/h/heli_deb.cfm |
| <i>Heliotropium arborescens</i> | forests, shrubs | http://eol.org/pages/487493/data ; http://www.pfaf.org/user/Plant.aspx?LatinName=Heliotropium+arborescens |
| <i>Iris domestica</i> | agricultural areas, shrubs, forests, grasslands, rock and scree, ruderal areas | http://www.signa.org/index.pl?Iris-domestica ; http://eol.org/pages/491571/data ; http://www.efloras.org/florataxon.aspx?flora_id=620&taxon_id=200028145 ; http://www.efloras.org/florataxon.aspx?flora_id=1&taxon_id=200028145 |
| <i>Isotoma axillaris</i> | rock and scree | http://www.anbg.gov.au/gnp/interns-2008/isotoma-spp.html |
| <i>Lilium formosanum</i> | grasslands, dunes, rock and scree | http://eol.org/pages/1083765/data ; http://www.efloras.org/florataxon.aspx?flora_id=3&taxon_id=200027716 ; http://www.pfaf.org/user/Plant.aspx?LatinName=Lilium+formosanum |
| <i>Lobelia inflata</i> | shrubs, forests, grasslands, agricultural areas, ruderal areas, wetland | http://eol.org/pages/593247/data ; http://www.herbs2000.com/homeopathy/lobelia.htm |
| <i>Monarda punctata</i> | grasslands, shrubs, forests, ruderal areas, dunes | http://eol.org/pages/579702/details ; http://www.floridata.com/ref/m/mona_pun.cfm ; http://www.missouribotanicalgarden.org/PlantFinder/PlantFinderDetails.aspx?kempercode=e290 ; http://www.pfaf.org/user/Plant.aspx?LatinName=Monarda+punctata |
| <i>Pennisetum macrourum</i> | riverine vegetation, grasslands, ruderal areas, wetlands, agricultural areas | http://eol.org/pages/1115490/data ; http://keyserver.lucidcentral.org |
| <i>Petunia integrifolia</i> | ruderal areas | http://alienplantsbelgium.be/content/petunia-integrifolia |

| | | |
|------------------------------|--|--|
| <i>Rudbeckia fulgida</i> | riverine vegetation, shrubs, forests, wetland, dunes | http://eol.org/pages/486458/details ; http://eol.org/pages/486458/data ; http://www.finegardening.com/orange-coneflower-rudbeckia-fulgida-var-sullivantii-goldsturm ; http://www.missouribotanicalgarden.org/PlantFinder/PlantFinderDetails.aspx?kempercode=g630 |
| <i>Rudbeckia triloba</i> | grasslands, ruderal areas, riverine vegetation, shrubs, forests, rock and scree, agricultural areas, wetland | http://eol.org/pages/467876/details , http://eol.org/pages/467876/data ; http://www.missouribotanicalgarden.org/PlantFinder/PlantFinderDetails.aspx?kempercode=b937 http://eol.org/pages/486458/details ; http://eol.org/pages/486458/data ; http://www.finegardening.com/orange-coneflower-rudbeckia-fulgida-var-sullivantii-goldsturm ; http://www.missouribotanicalgarden.org/PlantFinder/PlantFinderDetails.aspx?kempercode=g630 V; http://www.finegardening.com/brown-eyed-susan-rudbeckia-triloba ; http://www.missouriplants.com/Yellowalt/Rudbeckia_triloba_page.html |
| <i>Solidago ptarmicoides</i> | rock and scree, grasslands, dunes | http://eol.org/pages/482343/data ; http://www.efloras.org/florataxon.aspx?flora_id=1&taxon_id=242417294 ; http://www.thismia.com/S/Solidago_ptarmicoides.html |
| <i>Verbena rigida</i> | agricultural areas, shrubs, forests, grasslands, ruderal areas, riverine vegetation | http://eol.org/pages/579840/details ; http://eol.org/pages/579840/data ; http://www.issg.org/database/species/ecology.asp?ft=1&sts=EN&lang=EN&si=1371 |

Table S4: Climate models and modelling groups that provided data on future climate scenarios via the CORDEX portal (<https://www.cordex.org>). RCM = regional climate model; GCM = global climate model; GES = global emission scenario; ΔT = difference of mean annual temperature (2030-2080) and current climate (1950 – 2000); ΔP = increase of mean annual precipitation (2030-2080) relative to current climate. Values were averaged across the study region.

| Institute | RCM | Resolution (deg) | GCM | GES | ΔT (°C) | ΔP (%) |
|---|------------|-------------------------|------------|------------|-----------------------------------|----------------------------------|
| Swedish Meteorological and Hydrological Institute | RCA4 | 0.11 | EC-EARTH | RCP 2.6 | +1.1 | +2.1 |
| Swedish Meteorological and Hydrological Institute | RCA4 | 0.11 | CNRM-CM5 | RCP 4.5 | + 1.4 | +5.7 |
| Danish Meteorological Institute | HIRHAM5 | 0.11 | EC-EARTH | RCP 8.5 | + 2.4 | +5.0 |

Table S5: Evaluation statistics of species distribution models for the 15 species studied. Reported are mean TSS for all replicates and percentage of replicates that have a $TSS < 0.5$ (i.e. poor model performance), respectively, for each modelling technique. The minimum number of occurrence records per species was 50 (see Table 1 in Appendix). Abbreviations: CTA = Classification Tree Analysis; GBM = Generalized Boosting Model or usually called Boosted Regression Trees; RF = Random Forest; GLM = Generalized Linear Model; GAM = Generalized Additive Model; MARS = Multiple Adaptive Regression Splines; FDA = Flexible Discriminant Analysis; ANN = Artificial Neural Network;

| MODEL SPECIES | CTA | | GBM | | RF | | GLM | | GAM | | MARS | | FDA | | ANN | |
|---------------------------------|----------|---------|----------|---------|----------|---------|----------|---------|----------|---------|----------|---------|----------|---------|----------|---------|
| | Mean TSS | TSS<0.5 | Mean TSS | TSS<0.5 | Mean TSS | TSS<0.5 | Mean TSS | TSS<0.5 | Mean TSS | TSS<0.5 | Mean TSS | TSS<0.5 | Mean TSS | TSS<0.5 | Mean TSS | TSS<0.5 |
| <i>Amaranthus tricolor</i> | 0.487 | 37 | 0.640 | 10 | 0.643 | 17 | 0.566 | 0 | 0.494 | 67 | 0.613 | 10 | 0.547 | 33 | 0.583 | 17 |
| <i>Centaurea americana</i> | 0.744 | 0 | 0.858 | 0 | 0.877 | 0 | 0.828 | 0 | 0.788 | 0 | 0.835 | 0 | 0.822 | 0 | 0.744 | 7 |
| <i>Helianthus debilis</i> | 0.791 | 0 | 0.868 | 0 | 0.883 | 0 | 0.840 | 0 | 0.833 | 0 | 0.815 | 0 | 0.801 | 0 | 0.822 | 0 |
| <i>Heliotropium arborescens</i> | 0.693 | 7 | 0.792 | 0 | 0.794 | 0 | 0.662 | 0 | 0.817 | 0 | 0.745 | 3 | 0.714 | 3 | 0.642 | 7 |
| <i>Iris domestica</i> | 0.810 | 0 | 0.883 | 0 | 0.890 | 0 | 0.886 | 0 | 0.835 | 0 | 0.838 | 0 | 0.826 | 0 | 0.797 | 3 |
| <i>Isotoma axillaris</i> | 0.916 | 0 | 0.969 | 0 | 0.974 | 0 | 0.968 | 0 | 0.963 | 0 | 0.969 | 0 | 0.932 | 0 | 0.924 | 7 |
| <i>Lilium formosanum</i> | 0.837 | 0 | 0.914 | 0 | 0.925 | 0 | 0.896 | 0 | 0.911 | 0 | 0.920 | 0 | 0.884 | 0 | 0.878 | 0 |
| <i>Lobelia inflata</i> | 0.920 | 0 | 0.961 | 0 | 0.9600 | 0 | 0.927 | 0 | 0.920 | 0 | 0.928 | 0 | 0.916 | 0 | 0.914 | 0 |
| <i>Monarda punctata</i> | 0.855 | 0 | 0.933 | 0 | 0.946 | 0 | 0.938 | 0 | 0.960 | 0 | 0.912 | 0 | 0.921 | 0 | 0.899 | 0 |
| <i>Pennisetum macrourum</i> | 0.771 | 0 | 0.834 | 0 | 0.854 | 0 | 0.818 | 0 | 0.822 | 0 | 0.807 | 0 | 0.793 | 0 | 0.793 | 0 |
| <i>Petunia integrifolia</i> | 0.800 | 0 | 0.894 | 0 | 0.896 | 0 | 0.878 | 0 | 0.859 | 0 | 0.876 | 0 | 0.879 | 0 | 0.825 | 0 |
| <i>Rudbeckia fulgida</i> | 0.877 | 0 | 0.930 | 0 | 0.951 | 0 | 0.919 | 0 | 0.924 | 0 | 0.907 | 0 | 0.909 | 0 | 0.882 | 0 |
| <i>Rudbeckia triloba</i> | 0.940 | 0 | 0.967 | 0 | 0.969 | 0 | 0.918 | 0 | 0.915 | 0 | 0.939 | 0 | 0.927 | 0 | 0.918 | 0 |
| <i>Solidago parvicooides</i> | 0.709 | 3 | 0.887 | 0 | 0.892 | 0 | 0.800 | 0 | 0.910 | 0 | 0.831 | 0 | 0.843 | 0 | 0.770 | 7 |
| <i>Verbena rigida</i> | 0.879 | 0 | 0.920 | 0 | 0.927 | 0 | 0.883 | 0 | 0.905 | 0 | 0.874 | 0 | 0.841 | 0 | 0.895 | 0 |

Table S6: Linear regression of the number of cells simulated to be occupied in 2090 under the ‘low dispersal’ parameter setting on climate change scenario, cultivation frequency, and their interaction. Estimates of RCP-scenarios represent relative differences to the results obtained under constant climatic conditions. Lower AIC (Akaike Information Criterion) values indicate better model fits.

| Predictors | Estimate | Std. error | <i>p</i> -value | AIC | R ² |
|---|----------|------------|-----------------|------|----------------|
| Climate change scenario * Cultivation frequency | | | | 5538 | 0.65 |
| Baseline | 0.16 | 0.02 | <0.001 | | |
| RCP 2.6 | -0.34 | 0.03 | <0.001 | | |
| RCP 4.5 | 0.06 | 0.03 | 0.040 | | |
| RCP 8.5 | -0.34 | 0.03 | <0.001 | | |
| Cultivation frequency | 0.88 | 0.02 | <0.001 | | |
| RCP 2.6 : Cultivation frequency | -0.25 | 0.03 | <0.001 | | |
| RCP 4.5 : Cultivation frequency | 0.07 | 0.03 | 0.020 | | |
| RCP 8.5 : Cultivation frequency | -0.23 | 0.03 | <0.001 | | |
| excluding | | | | | |
| Climate change scenario | | | | 5982 | 0.60 |
| Cultivation frequency | | | | 8741 | 0.03 |
| Climate change scenario : Cultivation frequency | | | | 5703 | 0.64 |

Table S7: Assignment of habitat suitability to CORINE land cover classes (CLC) for the 15 species studied. For further information see CORINE land cover technical guide (<http://www.eea.europa.eu/>). CORINE land cover classes were reclassified to coarser habitat types (column ‘habitat’) and suitability classes assigned to each combination of habitat type and species (0 = not suitable, 10 = 10% of the area is suitable, 50 = 50% of the area is suitable, 100 = 100% of the area is suitable). Assignments were based on information sources listed in Supplementary Table S4.

| CLC | habitat | <i>Amaranthus tricolor</i> | <i>Centaurea americana</i> | <i>Helianthus debilis</i> | <i>Heliotropium arborescens</i> | <i>Iris domestica</i> | <i>Isotoma axillaris</i> | <i>Lilium formosanum</i> | <i>Lobelia inflata</i> | <i>Monarda punctata</i> | <i>Pennisetum macrourum</i> | <i>Petunium integrifolia</i> | <i>Rudbeckia fulgida</i> | <i>Rudbeckia triloba</i> | <i>Solidago parviflora</i> | <i>Verbena rigida</i> |
|--|---------------------|----------------------------|----------------------------|---------------------------|---------------------------------|-----------------------|--------------------------|--------------------------|------------------------|-------------------------|-----------------------------|------------------------------|--------------------------|--------------------------|----------------------------|-----------------------|
| 211, 212, 213, 221, 222, 223, 241, 242, 243, 244 | agricultural areas | 10 | 0 | 0 | 0 | 10 | 0 | 0 | 10 | 0 | 10 | 0 | 0 | 10 | 0 | 10 |
| 231, 321 | grasslands | 0 | 100 | 0 | 0 | 100 | 0 | 100 | 100 | 100 | 100 | 0 | 0 | 100 | 100 | 100 |
| 311, 312, 313, 324 | forest | 0 | 0 | 0 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100 |
| 322, 323 | shrubs | 0 | 100 | 0 | 100 | 100 | 0 | 0 | 100 | 100 | 0 | 0 | 100 | 100 | 0 | 100 |
| 331 | dunes | 0 | 0 | 50 | 0 | 0 | 0 | 50 | 0 | 50 | 0 | 0 | 50 | 0 | 50 | 0 |
| 332, 333 | rock and scree | 0 | 10 | 0 | 0 | 10 | 10 | 10 | 0 | 0 | 0 | 0 | 0 | 10 | 10 | 0 |
| 411, 412, 422, 423 | wetland | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100 | 0 | 100 | 0 | 100 | 100 | 0 | 0 |
| 421 | salt marshes | 0 | 0 | 0 | 0 | 0 | 0 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 511, 522 | riverine vegetation | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 50 | 0 | 50 | 50 | 0 | 50 |

Table S8: Estimated proportional area available for ornamental plant cultivation for land cover classes of CORINE (CLC). Whether gardens are included in a specific land cover class has been extracted from CLC technical guide (<http://www.eea.europa.eu/>). CLC classes not shown here include no specific plant cultivation areas for ornamentals. Data was taken from ref. 9.

| Class | Specification | area [%] |
|-------|--|----------|
| 111 | Continuous urban fabric | 5 |
| 112 | Discontinuous urban fabric | 5 |
| 121 | Industrial or commercial units | 0.1 |
| 122 | Road and rail networks and associated land | 0.1 |
| 141 | Green urban areas | 10 |
| 142 | Sport and leisure facilities | 0.1 |
| 211 | Non-irrigated arable land | 0.1 |
| 222 | Fruit trees and berry plantations | 0.1 |
| 242 | Complex cultivation patterns | 0.1 |
| 243 | Land principally occupied by agriculture, with significant areas of natural vegetation | 0.1 |

Table S9: Assumed area covered by individual/shoot of a species, assumed maximal fraction of a grid cell (= 250 x 250 m²) that can be colonized by each species, and calculated maximal carrying capacity (CC) in suitable sites.

| Species | Area individual [cm ²] | Cover value [%] | CC |
|---------------------------------|------------------------------------|-----------------|---------|
| <i>Amaranthus tricolor</i> | 1000 | 10 | 62500 |
| <i>Iris domestica</i> | 1000 | 10 | 62500 |
| <i>Centaurea americana</i> | 1000 | 20 | 125000 |
| <i>Helianthus debilis</i> | 1000 | 50 | 312500 |
| <i>Heliotropium arborescens</i> | 1000 | 20 | 125000 |
| <i>Isotoma axillaris</i> | 100 | 20 | 1250000 |
| <i>Lilium formosanum</i> | 100 | 10 | 625000 |
| <i>Lobelia inflata</i> | 1000 | 10 | 62500 |
| <i>Monarda punctata</i> | 1000 | 10 | 62500 |
| <i>Pennisetum macrourum</i> | 50 | 20 | 2500000 |
| <i>Petunia integrifolia</i> | 100 | 20 | 1250000 |
| <i>Rudbeckia fulgida</i> | 1000 | 15 | 93750 |
| <i>Rudbeckia triloba</i> | 1000 | 15 | 93750 |
| <i>Solidago ptarmicoides</i> | 1000 | 15 | 93750 |
| <i>Verbena rigida</i> | 100 | 10 | 625000 |

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