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Do non-native and dominant native species carry a similar risk of invasiveness? A case study for plants in Turkey

Ayşe Yazlık[1], Didem Ambarlı[2]

1 *Department of Plant Protection, Faculty of Agriculture, Düzce University, Düzce, Turkey* **2** *Department of Biological Sciences, Middle East Technical University, Ankara, Turkey*

Corresponding author: Ayşe Yazlık (ayseyazlik@gmail.com)

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Abstract

Most risk analysis studies in invasion biology have focused on the invasiveness of non-native species, even though some native species also can pose a high risk to the environment and human well-being. This is especially true under current global change, which may cause dominant native species to expand their range of distribution and have substantial effects on the ecosystem. In this study, the risk of invasiveness of five non-native and five native plant species in Turkey was evaluated using a standard risk screening protocol. All ten species selected for screening are known to be invasive in several parts of the world, i.e. non-native *Ailanthus altissima*, *Cuscuta campestris*, *Phytolacca americana*, *Robinia pseudoacacia* and *Sicyos angulatus*, and native *Cirsium arvense*, *Hedera helix*, *Onopordum acanthium*, *Phragmites australis* and *Sorghum halepense*. The Australian Weed Risk Assessment decision-support tool adapted to Turkey's geographical and climatic conditions was used for screening the study species based on their biological traits, ecology and management approaches. All species were classified as high-risk, with *R. pseudoacacia* among non-natives and *P. australis* among natives achieving the highest scores followed by *S. halepense*, *C. campestris*, *C. arvense*, *O. acanthium*, *P. americana*, *S. angulatus*, *A. altissima* and *H. helix*. Based on their risk scores, all non-native species were classified as invasive and all native species as 'expanding' for Turkey. An ordination based on the risk scores showed similarities between invasive and expanding species. The outcomes of this study indicate that species can have several risk-related traits resulting in high risk scores irrespective of their origin. Such species can modify their environment and interact with other species with severe consequences for biodiversity. It is argued that dominant species with highly negative environmental and socioeconomic impacts in their habitats should be included in priority lists for management measures irrespective of their origin (i.e. native or non-native). More studies are needed to evaluate the magnitude and prevalence of the present findings for other regions worldwide.

Keywords
Alien species, expansion, invasion, management, risk screening, Türkiye

Introduction

In the last decades, increased travel, trade and tourism in connection with globalisation and human population expansion have facilitated the deliberate and/or unintentional transport of plant and animal species beyond their natural biogeographical barriers (Hulme 2009; Şekercioğlu et al. 2011; Pyšek et al. 2017, 2020; Essl et al. 2019; Zenni et al. 2021). This has resulted in the introduction of non-native invasive species into new regions with consequent negative environmental and socioeconomic impacts (Pyšek et al. 2020). Under the challenging conditions of global change, a major task of invasion biology is to identify those high-risk species that are more likely to cause negative impacts. Usually, species that are either non-native invasive or 'expanding native' (Simberloff et al. 2012; Díaz et al. 2019; Essl et al. 2019; Simberloff 2022; Yazlık and Üremiş 2022) and that become dominant in natural habitats may exert direct or indirect impacts on community structure and composition, species interactions and ecosystem functions, all of which can result in a 'domino effect' (Hawkins et al. 2015; Pyšek et al. 2017, 2020; Hulme and Bernard-Verdier 2018; Díaz et al. 2019; Brundu et al. 2020).

Identifying species posing a high risk of invasiveness is sometimes challenging due to knowledge gaps in their biology/ecology, and this represents a limitation for the implementation of effective management and control measures (Hulme 2009; Hulme and Bernard-Verdier 2018; Yazlık et al. 2018; Pyšek et al. 2020). This is a crucial aspect in risk screening/identification (i.e. the first step in risk analysis followed by risk assessment,risk management and communication: e.g. Vilizzi et al. 2022) especially given current debate on whether non-native species can be considered as a contribution to the biodiversity of the invaded regions (Simberloff 2011; Simberloff et al. 2012; Pauchard et al. 2018), hence in contrast to evidence for their environmental and socioeconomic impacts (Hawkins et al. 2015; Nentwig et al. 2016; Rumlerová et al. 2016; Bacher et al. 2018; Yazlık et al. 2018; Starfinger and Schrader 2021). For this reason, the first step in the identification of potentially high-risk invasive species is to find out their native or non-native status in the regions where they are found (e.g. Uludağ et al. 2017) and then determine their potential environmental and socioeconomic impacts (Hawkins et al. 2015; Bacher et al. 2018; Pauchard et al. 2018; Tanner and Fried 2020; Yazlık and Albayrak 2020; EPPO 2021). This provides for an opportunity to select those species more likely to be selected for risk analysis (Hawkins et al. 2015; Nentwig et al. 2016; Rumlerová et al. 2016; Bacher et al. 2018; Yazlık et al. 2018).

An effective means to identify high-risk invasive species is by the use of risk screening decision-support tools (see Vilizzi et al. 2022). These allow to carry out follow-up risk assessment after identification of the species classified as carrying a high risk of invasiveness for a certain risk assessment area (Díaz et al. 2019; Lenzner et al. 2019; Pyšek et al. 2020). At the same time, the drivers of global change, such as climate and land-use activities and accessibility, can also cause an increase in the range of expansion and abundance of the native species, which are then referred to as expanding species. Some examples are the expansion of tall grass plants in the absence of large herbivores (Corazza et al. 2016), liana infestations in tropical forests following disturbance (Schnitzer and Bongers 2011), and graminoids and shrubs expanding in tundra as a result of climate change (McManus et al. 2012).

Several mechanisms including the availability of free niches and increased competitive ability are involved in the invasion process by non-native species (Catford et al. 2009; Hiero and Callaway 2021). Yet, several plant species within their native range behave like invasive plants (Pyšek et al. 2004; Simberloff 2011; Simberloff et al. 2012; Hejda et al. 2021; Yazlık and Üremiş 2022). Although there is an ongoing debate as to whether the impacts of non-native invasive plants differ from those of expanding native (dominant) plants (Simberloff et al. 2012, 2013; Hejda et al. 2021), there is solid evidence that both non-native species' invasions and the spread of dominant native species may pose threats to biodiversity and sustainability (Hejda et al. 2021; Yazlık and Üremiş 2022). This is also because native dominant plant species are likely to be invasive outside their native range (Pyšek et al. 2009; Phillips et al. 2010; Hejda et al. 2021). However, there are very few native dominant plant species that have been compared with non-native invasive plant species in terms of their fast spread and negative impacts on vegetation (e.g. Hejda 2013). It is therefore argued that dominant expanding species should be evaluated in a similar way to non-native species by risk analysis in order to understand the threats they may pose to the ecosystem (Sohrabi et al. 2020; Jan et al. 2022). Importantly, identifying potential invasion/expansion of these species by risk analysis will play an important role in preventing/mitigating environmental and socioeconomic impacts, especially in terms of biodiversity loss.

The aim of this study was to show that some dominant native plant species can pose a high risk of invasiveness as much as non-native plant species using a dataset from Turkey. To this end, a risk screening was conducted on ten plant species in Turkey that are registered as non-native invasive in several geographical regions worldwide. The specific objectives were to: (i) determine the invasion/expansion status of the study species in Turkey, and (ii) search for a relationship between the risk status of these species and their origin. The purpose of this study is to emphasise the necessity of approaching expanding species from an invasiveness perspective.

Methods

Species selection

Four criteria were used for selection of the plant species for screening. Firstly, species were selected that have a wide distribution in three biogeographic regions of Turkey, namely the Euro-Siberian, Iran-Turanian and Mediterranean (Bizim Bitkiler 2020). Secondly, species were selected for which no risk analysis studies have been conducted in Turkey, but are defined as non-native invasive plants in different parts of the world

Table 1. Information on the species screened for their risk of invasiveness in Turkey. EPPO code: code used for plant taxa by the European and Mediterranean Plant Protection Organization.

Species	Family	Origin	Lifetime and form	EPPO code
Non-native				
Ailanthus altissima (Mill.) Swingle	Simaroubaceae	China	Perennial tree	AILAL
Cuscuta campestris Yunck.	Convolvulaceae	America	Parasitic; climbing annual or perennial herb	CVCCA
Phytolacca americana L.	Phytolaccaceae	America	Polycarpic perennial herb	PHTAM
Robinia pseudoacacia L.	Fabaceae	America	Perennial tree	ROBPS
Sicyos angulatus L.	Cucurbitaceae	America	Climbing or creeping annual herb	SIYAN
Native				
Cirsium arvense (L.) Scop.	Asteraceae	Turkey	Polycarpic perennial herb	CIRAR
Hedera helix L.	Araliaceae	Turkey	Climbing or creeping perennial woody	HEEHE
Onopordum acanthium L.	Asteraceae	Turkey	Annual or biennial herb	ONRAC
Phragmites australis (Cav.) Trin. ex Steud.	Poaceae	Turkey	Perennial herb	PHRCO
Sorghum halepense (L.) Pers.	Poaceae	Turkey	Perennial herb	SORHA

(GISD 2022). Of note, this type of selection has been proposed for studies comparing invasive non-native species with native species (van Kleunen et al. 2010). These first two criteria enabled the selection of species with a high potential for impacts whilst their risk of invasiveness was not known beforehand. Thirdly, species were selected that have biological traits of invasiveness. To this end, the species' life-history, biological, morphological and physiological traits were evaluated and the following were considered: adaptation to different habitats, soil type, pH range, competitive abilities, presence of below- and above-ground structures, and high generative and/or vegetative capacity. Lastly, species were selected that have high environmental and socioeconomic impacts, such as negative effects on natural vegetation, allelochemical contents, and toxic and/or injurious to humans and animals (Yazlık et al. 2017; Yazlık et al. 2018; Yazlık and Albayrak 2020; Aksan and Yazlık 2021). Conducting a risk analysis on non-native species and determining their invasiveness status was suggested in previous studies for Turkey (Uludağ et al. 2017; Yazlık et al. 2018; Yazlık 2022). Notably, the study species were not limited to pairs of native and non-native species with certain traits or habitat features, which would make drawing generalisable conclusions more difficult. On the contrary, the objective was to select species with a similar level of invasiveness but different origin and habitat. As a result, five non-native and five native species were selected: *Ailanthus altissima*, *Cuscuta campestris*, *Phytolacca americana*, *Robinia pseudoacacia* and *Sicyos angulatus* as non-native, and *Cirsium arvense*, *Hedera helix*, *Onopordum acanthium*, *Phragmites australis* and *Sorghum halepense*as as native (Table 1).

Risk screening

For risk screening, a decision-support tool adapted from the Australian Weed Risk Assessment (WRA: Pheloung et al. 1999) was used accounting for the geographical and climatic conditions of Turkey, namely the Türkiye Weed Risk Assessment: TR-WRA (Suppl. material 1: Table S1). The screening protocol for the TR-WRA involves 49 questions dealing with the species' biological traits, environmental impacts and management planning. The following modifications were done to the original set of questions (Qs): (i) 'suitability of the species to Australian climate' was changed to 'suitability of the species to the climate in Turkey' (Q 2.1); (ii) 'native or naturalised in regions with extended dry periods' was changed to 'native or naturalised in regions with a mild climate' (Q 2.4); (iii) 'presence of effective natural enemies in Australia' was changed to 'presence of effective natural enemies in Turkey' (Q 8.5). For each answered question, the species is assigned a score between –2 and 2, and the Q-specific scores are then summed to produce a total risk score (RS), which ranges from a minimum of –14 to a maximum of 29. However, in the question about the quality of climate matching data (Q 2.2), as all screened species scored high (i.e. with 2 points) and their natural ranges are well known (Table 1), the scores were not included in the RS, and these scores were not shown in the risk analysis table (Suppl. material 1: Table S1). In addition, 'no' or 'unknown' was added to the choice of some questions that were not related to the study species or for unknown risks (Suppl. material 1: Table S1).

As no RS thresholds for invasiveness identification were set by the authors who designed the protocols for the A-WRA test (Pheloung et al. 1999; Andreu and Vilà 2010), after accounting for similar risk outcomes scoring higher than the maximum value (e.g. Morais et al. 2017), the RS was modified to being ≥ 29. Also, at least ten answers are required for the evaluation of a species (Andreu and Vilà 2010; Morais et al. 2017). Overall, following Andreu and Vilà (2010), the TR-WRA scoring system can be used to classify species into three groups according to their level of risk: (i) species' occurrence in the risk assessment area acceptable (score < 1); (ii) species introduction in the risk assessment area prohibited (score > 6); or (iii) further work needed for a reliable risk screening outcome (score between 1 and 6). If a native species is identified in the second group, this implies that species management is required.

Data collection and statistical analysis

The information required to answer each question was obtained from national and international literature. Search for literature was conducted in Google Scholar, Web of Science, Scopus and ULAKBİM (Suppl. material 1: Table S2). In addition, a monograph (Tanner and Fried 2020), data sheets (EPPO 2010, 2021), one 'grey literature' reference (Köstekçi 2010), and online databases (European Project DAISIE: http://www.europe-aliens.org/; USDA Plants database http://plants.usda.gov; International Survey of Herbicide Resistant Weeds: http://www.weedscience.org; Global Invasive Species Database: http://www.issg.org/database) were used.

Multiple Correspondence Analysis (MCA), which is suitable for ordination of categorical data (Abdi and Williams 2010), was employed to visualise variation in the species' risk scores and their relationship with the species' origin (i.e. non-native vs native), and to identify similar species in terms of scores. The output of MCA can be interpreted similar to a quantitative ordination, with species closer to each other having higher similarities in their scores. Function *mca* of package *FactoMineR* (Le et al. 2008) was used to implement MCA in R version 3.5.0 (R Core Team 2020). Before analysis, the scores for a total of 17 questions with the same scores for all species were omitted from the dataset as they did not carry useful information for an ordination. These questions included: naturalised or not (Q 1.2 in Suppl. material 1: Table S1), climatic suitability (Q 2.1), climate match (Q 2.2), environmental versatility (Q 2.3), repeated introductions (Q 2.5), garden/amenity/disturbance weed (Q 3.2), agricultural weed (Q 3.3), environmental weed (Q 3.4), allelopathy (Q 4.2), host for unwanted species (Q 4.6), plant of infertile soils (Q 4.10), geophyte (Q 5.4), reproductive failure (Q 6.1), viable seeds (Q 6.2) pollinator requirement (Q 6.5), unintentional dispersion (Q 7.1) and prolific seed production (Q 8.1). Furthermore, Q 6.4 with 'unknown' as an answer was removed from the dataset.

Results

Following risk screening, all ten species were found to carry a high risk of invasiveness (Suppl. material 1: Table S1). The species with the highest scores were *R. pseudoacacia* (RS = 32) among non-natives and *P. australis* (40) among natives, followed by *S. halepense* (33), *C. arvense* and *C. campestris* (31), *O. acanthium* and *P. americana* (30), *S. angulatus* (29), *A. altissima* and *H. helix* (28). Based on these scores, all non-native species were risk-ranked as invasive and all native species as expanding for Turkey. All species were recorded in various habitats, predominantly agricultural but also sandy, saline, rocky and ruderal (Table 2).

Although these species have very different characteristics from each other, similar scores were achieved in the sub-categories related to their dominant characters. For example, when the dispersal mechanism (Section 7: Suppl. material 1: Table S1) was analysed, the total score range of the species changed between 5 and 8 according to the character-

Table 2. Habitats in Turkey of the species under study (for evidence, see Section 3 in Suppl. material 1: Table S1).

Habitat	*A. altissima*	*C. arvense*	*C. campestris*	*H. helix*	*O. acanthium*	*P. americana*	*P. australis*	*R. pseudoacacia*	*S. angulatus*	*S. halepense*
Arable	*	*	*	*	*	*	*	*	*	*
Dryland	-	*	*	-	*	-	-	-	-	-
Forest	*	-	-	*	-	*	*	*	*	*
Grassland	*	*	*	-	*	*	*	*	-	*
Riparian	*	*	*	*	*	*	*	*	*	*
Rocky	*	*	*	*	*	-	*	*	-	*
Ruderal	*	*	*	*	*	*	*	*	*	*
Saline	*	*	*	*	*	*	*	*	-	*
Sandy	*	*	*	*	*	*	*	*	-	*

istics of the plant (Q 7.4: Suppl. material 1: Table S1). In addition, the grading of questions or the absence of a certain feature affected the total score. For example, while *C. arvense*, *O. acanthium*, *P. australis* and *R. pseudoacacia* achieved the highest score of 2 in this section, *H. helix* and *S. halepense* achieved a score of −1, and other species were scored in the range of 0 to 1. Thus, a species that is known to have definite spread by wind and a species that is likely to drift to a limited area in very strong winds are not given the same score. Although the question-specific risk scores for the native plant species were mostly either negative or 0 because of their origin, this did not affect their total (high-risk) score. For example, since natural enemies of *C. arvense* and *O. acanthium* are in a limited range and not used as biological control agents, both plants scored −1 instead of the lowest score of −2 for the question (Q 8.5) related to the presence of natural enemies.

The two dimensions of the MCA analysis explained ≈39% of the variation in the data (Fig. 1; Suppl. material 1: Table S3). Amongst the 69 attributes included, those with the highest contribution to the first axis of variation were absence of natural enemies and naturalisation outside the native range (Suppl. material 1: Table S4). For the second axis, those variables were no wind dispersal and properties of propagule banks (Suppl. material 1: Table S4 and Fig. S1). The first axis represented a gradient from two native species with no naturalisation but natural enemies (i.e. *C. arvense* and *O. acanthium*) to non-native species, which were all located in the negative part of the second axis. Non-native *C. campestris* was very close to the above two natives, indicating similarity in scores. The three other native species (i.e. *H. helix*, *P. australis* and *S. halepense*) were located far from the other natives along the secondary axis, indicating weak similarity. Based on the answers to the risk screening questionnaire, native and non-native species were not clearly grouped in the ordination space. Overall, MCA showed that similarity in terms of risk can be high amongst species of native and non-native species and low between two native species.

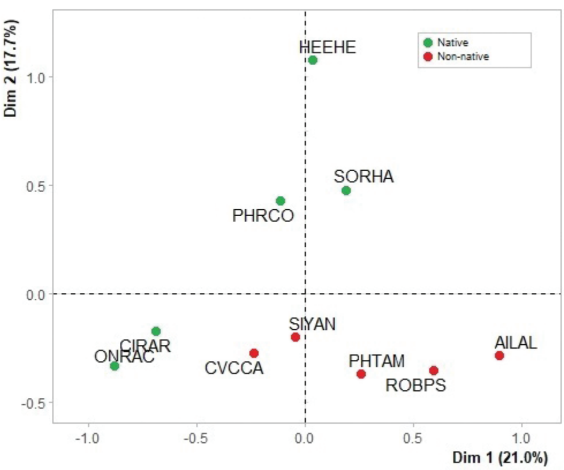

Figure 1. Multiple correspondence analysis factor map of the 10 plant species screened for their risk of invasiveness for Turkey according to their risk scores. Species labelled with their EPPO codes (see Table 1).

Discussion

This study has shown similar risk levels for non-native and native species with high potential of exerting negative impacts on both ecosystems and human well-being. In addition, this study is the first to provide a dataset of national evaluation for Turkey on the invasion/expansion status of ten dominant plant species that are registered as non-native invasive plants in different geographies (GISD 2022), but whose risk status has so far been unknown in this country. The present results showed that the invasion/expansion status of dominant plants may be independent of their local range, thereby emphasising the importance to evaluate species not only according to their biogeographical origin but also to their biological, morphological and physiological characteristics as well as environmental and socioeconomic impacts. Clearly, more studies relying on larger sample sizes are needed to quantify the magnitude and prevalence of this first evidence provided for Turkey.

The ten species under study were interpreted in two groups by accounting for their local distributional range, risk scores and human-induced dispersal. Accordingly, native species were considered as 'expanding apophytic', which are quite aggressive, spread rapidly and affect vegetation (Yücel et al. 2019; Hejda et al. 2021), whereas non-native species were considered as 'invasive anthropophytes' (*sensu* Yücel et al. 2019). All ten species have a dominant distribution in various habitats (i.e. agriculture, coastal, forest: Table 2) and human influence has a high share in their spread. In this respect, the most important factors are the 'weed' status of these species in agricultural habitats combined with their competitive abilities such as morphological characteristics (Yazlık and Tepe 2001; Kaçan and Boz 2015; Uludağ 2015; Üstüner et al. 2015; Sezer and Kolören 2019; Terzioğlu and Ergül Bozkurt 2020; Yazlık and Albayrak 2020; Aksan and Yazlık 2021). Specifically, clonal growth (Bímová et al. 2003), high biomass (van Kleunen et al. 2010; Hejda 2013; Canavan et al. 2019) and a large number of branches/tillerings (Hejda 2013; Yazlık and Üremiş 2022) were all traits associated with high invasiveness. In addition, serious problems have been reported regarding the presence of these species in their habitats (Table 2), which is a major reason for their high-risk scores, hence irrespective of their origin. Below, details are provided as to why these non-native and native species were found to carry a similar level of risk.

Native *O. acanthium* has negative impacts due to its superior competitiveness, spread and unpalatability based on its thorny structure, seed volatiles and re-sprout from root shoots, all of which cause vegetation degradation, decrease in agricultural production, injury in animals,deterioration of livestock nutrition, and labour costs (Pınar et al. 2018; Aksan et al. 2019; Aksan and Yazlık 2021). This is similar to native *C. arvense*, *P. australis* and *S. halepense*, which have dominant generative and vegetative propagation abilities (Suppl. material 1: Table S1). These species also create dense populations in habitats such as beaches and sand dunes, especially in agricultural and pasture lands, causing serious negative impacts on vegetation (Yazlık and Tepe 2001; Köstekçi 2010; Meyerson et al. 2010; Aksan et al. 2019; Aksan and Yazlık 2021; Erbaş and Doğan 2022; Jan et al. 2022; Yazlık and Üremiş 2022).

Hedera helix is present primarily in forests and urban habitats (Table 2) but also in agricultural habitats (e.g. nurseries, hazelnut orchards: Yazlık et al. 2019; Aksoy and Çelik 2020; Güneş Özkan et al. 2020). One of the main factors for the prevalence of this species in urban habitats is its use as an ornamental plant in parks or home gardens, while at the same time this species has a major impact on plant community composition in forests. *Cuscuta campestris*, *P. americana* and *S. angulatus* are naturalised non-native plants in Turkey that appear to occupy more than one habitat (Table 2). A parasitic plant with a wide host range, *C. campestris*, which is one of the species with the highest impacts worldwide (Yazlık et al. 2017), exerts major negative impacts by infecting cultivated plants in agricultural habitats, affecting rail ballast in railways, increasing fire risk, and being toxic to humans and animals (Yazlık and Albayrak 2020). Finally, non-native *P. americana* and *S. angulatus* are found in agricultural habitats that are generally considered to pose serious problems to agricultural production (Terzioglu and Ansin 1999; Korkmaz et al. 2016; Sezer and Kolören 2019).

The present risk screening study also determined the potential of non-native species to cause indirectly high risks in terms of plant diseases and nematode transmission in the areas where they are found (Suppl. material 1: Table S1). For example, *P. americana* is reported to provide suitable host conditions for five different nematodes (i.e. *Meloidogyne arenaria*, *M. floridensis*, *M. incognita*, *M. javanica* and *M. mayaguensis*: Kaur et al. 2007). Although there is no record of nematodes that are a problem for this species in Turkey, three nematodes reported by Kaur et al. (2007) are present in the country, namely *M. arenaria*, *M. incognita* and *M. javanica* (Özarslandan and Elekçioğlu 2010). Therefore, the interaction of *P. americana* with existing nematodes in the habitats of Turkey may create secondary problems by enhancing their further spread. This is especially important for arable lands, as there is evidence of damage by nematodes on cultivated plants (e.g. Özarslandan and Elekçioğlu 2010). Conversely, in terms of host or vector status of disease agents, *C. campestris* is a vector for virus and phytoplasma diseases (Yazlık and Albayrak 2020), whereas *S. angulatus* poses a high risk by being host to the watermelon mosaic virus (WMV-2: Korkmaz et al. 2016). Another example of a host is for *R. pseudoacacia*, which has the host status of *Viscum album* – a most problematic weed for many orchards in Turkey (Üstüner et al. 2015). Therefore, this non-native plant can contribute to the distribution of this parasitic plant.

Human-mediated dispersal was an important factor for the high risk of invasiveness identified in this study. Evidence shows that some of the screened species have often been reported as problematic weeds in agricultural areas and their prevalence may be due to their dispersal via contaminated agricultural tools and equipment with plant parts (Suppl. material 1: Table S1). Furthermore, transportation via road corridors can be an important channel for plant invasions (Lemke et al. 2021), as in the case of *C. campestris* (Yazlık and Albayrak 2020). Moreover, cultivation of *R. pseudoacacia*, which started 70 years ago in Turkey, is supported on the basis that it provides important socioeconomic benefits, such as erosion control, honey production with increased nectar provision, and timber use (BOEP 2013; Onur and Acar 2017). Therefore, the dispersal of some non-native plants, including *A. altissima*, can occur with direct human contribution due to their economic value. This is in agreement with the contextual assessment made by Vítková et al. (2020) in the decision to cultivate *R. pseudoacacia* in its non-native ranges. Therefore, the decision to continue the cultivation of high-risk non-native plants in Turkey as discussed in this study should be considered depending on the regional, ecological, conservation and socio-economic context.

The long-term presence of the study species were considered as another factor supporting their widespread distribution. For instance, *A. altissima*, *P. australis*, *R. pseudoacacia* and *S. halepense* not only in Turkey but also in several other regions worldwide is known to be widespread (POWO 2021). This was reflected by these species' high-risk scores because many species with long residence time are more likely to have a niche and geographic spread (Sychrová et al. 2022). At this stage, it should be taken into account that long residence time may also create problems in control studies of related species, even if native. For example, it has been reported that the herbicide Glyphosate applied at the edges of irrigation canals was not fully successful to combat *P. australis* in the Aydın plain, which is one of the most important polyculture crop production plains in the Aegean region of Turkey. This is because this species has a long-term persistent population in those ruderal habitats and integrated applications by mowing along the canal sides also cannot be made (Erbaş and Doğan 2022).

The species screened in this study are also affected by human activities (intentionally and/or unintentionally) besides spread and establishment in various habitats (Table 2). Amongst the different habitats, it has been emphasised that arable land is the most occupied by non-native plants, whereas natural and semi-natural grasslands are less invaded (Chytrý et al. 2008; Pyšek et al. 2009; Jauni and Hyvönen 2010). For instance, among the study species, *P. americana* has been reported from agricultural, forest and coastal habitats but as problematic especially in arable lands, due to shading and harvesting difficulties, such as for kiwi fruit (Sezer and Kolören 2019) and tea (Terzioğlu and Ergül Bozkurt 2020). Similarly, the screened native species have also been reported in several habitats including arable lands (Yazlık and Tepe 2001; Kaçan and Boz 2015; Yazlık and Üremiş 2022). For instance, the incidence of *P. australis* was determined as 48% and the density as 12 plants/m^2 in traditional vineyards of Manisa province in the Aegean Region of Turkey (Kaçan and Boz 2015).

Dominant native species can also cause demographic issues as a result of human-induced changes to the environment (Valéry et al. 2009; Simberloff 2011; Méndez et al. 2014; Sohrabi et al. 2020; Jan et al. 2022) thereby posing management challenges under current scenarios of global change (Simberloff 2011; Méndez et al. 2014). Nevertheless, *P. australis* (the native species with the highest risk score in this study) has also socioeconomic aspects on the country's trade and local people in the Sultan Marsh Nature Park, which is included in the List of Class A Wetlands in accordance to the second and third articles of the International Ramsar Convention in Turkey (Ramsar site no. 661 - https://rsis.ramsar.org/ris/661). Approximately 1500 tons of reeds (i.e. *P. australis* and *Typha* spp.) are cut annually by the local people in Sultan Marsh with most of the cut reeds being exported. The amount of thatch exported is approximately

300,000–400,000 bundles per year, and in 1995 a reed tying and storage facility was established in the town of Sindelhöyük. In addition, reeds (especially *P. australis*, which is a pure community represented by almost a single species in Yay Lake in the south and southwest areas of the Sultan marshes: Hamzaoğlu and Aksoy 2006) are used as roofing material (thatched roof) and animal feed in the region, where they represent an important source of income (Karadeniz 2000; Hamzaoğlu and Aksoy 2006; Sarısoy 2015). As a result, it is recommended that native (*P. australis*) and non-native (*A. altissima* and *R. pseudoacacia*) high-risk species with socioeconomic contributions should be monitored across Turkey and context-dependent prevention and management approaches should be developed in case of local adverse impacts.

The presence of natural enemies to native species is another important criterion to determine their risk of invasiveness (Q 8.5: Suppl. material 1: Table S1). For example, despite the existence of natural enemies for *O. acanthium* such as *Homoeosoma nebulellum* (Lepidoptera, Pyralidae) and *Larinus latus* (Coleoptera, Curculionidae) (Gültekin 2008; Yücel and Çobanoğlu 2016), the potential of these insects as biological control agents is limited (Gültekin 2008; Yücel and Çobanoğlu 2016). This is also true of *C. arvense*, whose natural enemies are recorded in its local distributional range (Kedici et al. 1994). Therefore, control of these plant species by such natural enemies may be limited to areas where these agents are present. For this reason, it is suggested that studies should be carried out to investigate the role of such natural enemies for an effective control and to identify related plant species in Turkey as biological control agents.

Due to their high risk of invasiveness, all species screened in this study (and regardless of their origin) should be listed as priority species. Sustainability of existing native species and reducing or stopping the negative impacts of invasive/expanding species can be possible by prevention. To achieve this objective, awareness-raising activities, training and effective species-specific management programmes (including the use of clean equipment in production areas, human-induced transportation of plant parts, Integrated Weed Management (IWM) application methods, and the use of non-native ornamental plants) should be organised based on the species' habitat. Effective management programmes are also important in terms of setting precautionary measures in plant transitions from Turkey to different geographies, as indicated by the large number of weed species originating from Turkey and being invasive or naturalised in different geographies/continents worldwide (A. Yazlık, unpublished data). To this end, implementation of effective biosecurity measures and cooperation amongst stakeholder groups would help in such efforts (Guo 2006; Lenzner et al. 2019; Pyšek et al. 2020; Wallingford et al. 2020; Yang et al. 2021).

Overall, if high-risk species disperse into areas other than their native habitats or geographic regions, additional risks may arise and the extent of the resulting impacts may increase. Further environmental and socioeconomic impacts can be expected in range-shifting non-native species due to hybridisation (Essl et al. 2019; Wallingford et al. 2020; Seebens et al. 2021). However, this requires an understanding of their potential interactions in new environments (Guo 2006; Wallingford et al. 2020; Seebens et al. 2021) as well as of the extent of such impacts (Wallingford et al. 2020; Simberloff 2022). All of this would require monitoring programmes and gathering local ecological information. For these reasons, it is believed that the present study can broaden the perspective about native and non-native species and add new data to the knowledge of related plants.

Conclusions

The present study has provided evidence for how both non-native and native species can result in high-risk scores of invasiveness independent of their native range. This suggests that further studies should be carried out on the extent and size of the impact exerted by such species. As research on invasiveness has been strongly focused on non-native species, it is hoped that the present study will point to the necessity of working on dominant native (expanding) species. Considering the results of the ten species investigated, it is suggested that further studies in risk analysis should include not only non-native species but also all dominant species that are known to cause high impacts. This is because damage to natural ecosystems is in most cases an irreversible process (Křivánek and Pyšek 2006; Brundu et al. 2020). Moreover, considering all aspects of socioeconomic and environmental changes at the national level provides a resource to monitor more effectively the potential developments of future biological invasions (Latombe et al. 2022). Therefore, it is suggested that invasive/expanding species lists should be created on a regional basis in view of risk analysis studies. At the same time, it is recommended that priority should be given to the establishment of management programmes (Brundu et al. 2020) and the implementation of effective biosecurity measures (Latombe et al. 2022) for species whose invasive/expanding status has been determined by risk analysis. Given the presence of the species screened in this study in different habitats across Turkey, appropriate management programmes should be implemented by taking into account the IWM principle. In particular, it is recommended that research institutes working on biological control in Turkey (e.g. Adana Biological Control Research Institute, which carries out studies on mass insect production) should consider the research on the natural enemies mentioned in this study. Finally, considering urban habitats, public awareness should be raised and decision-makers should be informed about the use of high-risk plants such as *A. altissima*, *H. helix* and *R. pseudoacacia*, which are sold and used as ornamental and/or landscape plants country-wide.

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Supplementary material 1

Tables S1–S3, Figure S1

Authors: Ayşe Yazlık, Didem Ambarlı

Data type: Tables and figure (docx. file)

Explanation note: **Table S1.** The risk analysis of the native and alien taxa. From left to right columns show question categories, questions and possible scores, scores for each non-native and native species, notes for yes/no. **Table S2.** List of references used for scoring the impact of the study species. **Table S3.** The proportion of variances retained by the dimensions of MCA. **Table S4.** Contribution of each variable to the MCA dimensions. Only the first five dimensions were presented. Variables indicated in Suppl. material 1: Table S1 are shortened in the first column. As the variables are categorical, their values with respect to a specific attribute are indicated with numbers at the end of variable labels. **Figure S1.** MCA ordination with variables. As the variables are categorical, their values with respect to a specific attribute are indicated with numbers at the end of variable labels. To prevent overlapping labels, small lines are used but still some attributes cannot be visualised due to overlaps.

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