Plant and Vegetation 14

Milan Chytrý
Jiří Danihelka
Zdeněk Kaplan
Petr Pyšek *Editors*

Flora and Vegetation of the Czech Republic

Springer

Milan Chytrý • Jiří Danihelka • Zdeněk Kaplan
Petr Pyšek
Editors

Flora and Vegetation of the Czech Republic

Springer

Editors
Milan Chytrý
Department of Botany and Zoology
Masaryk University
Brno, Czech Republic

Jiří Danihelka
Department of Botany and Zoology
Masaryk University
Brno, Czech Republic

Zdeněk Kaplan
Institute of Botany
The Czech Academy of Sciences
Průhonice, Czech Republic

Petr Pyšek
Institute of Botany
The Czech Academy of Sciences
Průhonice, Czech Republic

ISSN 1875-1318 ISSN 1875-1326 (electronic)
Plant and Vegetation
ISBN 978-3-319-63180-6 ISBN 978-3-319-63181-3 (eBook)
DOI 10.1007/978-3-319-63181-3

Library of Congress Control Number: 2017954304

Printed on acid-free paper

This Springer imprint is published by Springer Nature
The registered company is Springer International Publishing AG
The registered company address is: Gewerbestrasse 11, 6330 Cham, Switzerland

Chapter 8
Plant Invasions in the Czech Republic

Petr Pyšek, Milan Chytrý, Jan Pergl, Jiří Sádlo, and Jan Wild

Abstract At present there are 1454 alien taxa (species, subspecies, varieties, hybrids and cultivars) of vascular plants recorded in the Czech Republic, among them 350 archaeophytes, introduced since the beginning of the Neolithic until the end of the Middle Ages, and 1104 neophytes, introduced in the Modern Period. Of the total number, 985 (67.7%) taxa are classified as casual, 408 (28.1%) as naturalized but non-invasive and 61 (4.2%) as invasive. Aliens make up 33.1% of the total plant diversity recorded in this country, or 14.4% of the permanently present flora. The highest levels of invasion of plant communities are recorded in cities and villages and their surroundings, floodplains of large rivers, disturbed landscapes in the north, and agricultural landscapes and forest plantations in the warm lowlands, especially in southern, central and eastern Bohemia. The habitats and vegetation types harbouring the highest percentages of alien species in the Czech Republic are generally those with a high level of disturbance or with a fluctuating input of nutrients.

8.1 Introduction

Thorough studies of plants in human-made habitats have been a traditional topic of Czech botany since the 1970s (e.g. Hejný et al. 1973, 1979; Jehlík and Hejný 1974; Jehlík 1998). The systematic research into the biogeography and ecology of alien

P. Pyšek (✉)
Institute of Botany, The Czech Academy of Sciences, 252 43 Průhonice, Czech Republic

Department of Ecology, Faculty of Science, Charles University,
Viničná 7, 128 44 Praha 2, Czech Republic
e-mail: petr.pysek@ibot.cas.cz

M. Chytrý
Department of Botany and Zoology, Masaryk University,
Kotlářská 2, 611 37 Brno, Czech Republic
e-mail: chytry@sci.muni.cz

J. Pergl • J. Sádlo • J. Wild
Institute of Botany, The Czech Academy of Sciences, 252 43 Průhonice, Czech Republic
e-mail: jan.pergl@ibot.cas.cz; saadlo@volny.cz; jan.wild@ibot.cas.cz

M. Chytrý et al. (eds.), *Flora and Vegetation of the Czech Republic*, Plant and Vegetation 14, DOI 10.1007/978-3-319-63181-3_8

plants in this country was triggered by the publication of the *Catalogue of alien plants of the Czech Republic* (Pyšek et al. 2002b, updated by Pyšek et al. 2012b). In the last decade, a wide array of issues were addressed at the regional scale, such as the role of species traits in determining species invasiveness (e.g. Pyšek et al. 2009, 2011a; Kubešová et al. 2010; Moravcová et al. 2010), patterns of habitat invasibility (Chytrý et al. 2005, 2008a, 2009b; Pyšek et al. 2005; Simonová and Lososová 2008; Lososová and Cimalová 2009; Lososová and Grulich 2009), including invasions of nature reserves (Pyšek et al. 2002b, 2003a), as well as topics related to pathways of introduction (Pergl et al. 2016c), impact and risk assessment (Křivánek and Pyšek 2006; Chytrý et al. 2009b; Hejda et al. 2009a; Pyšek et al. 2011b, 2012c; Pergl et al. 2016d) and case studies of individual invasive species (see Pyšek et al. 2012a for an overview, and Hejda 2013; Horáčková et al. 2014; Čuda et al. 2015, 2016; Reif et al. 2016; Skálová et al. 2017; Vítková et al. 2017 for examples of recent research).

Here we (i) review patterns in the diversity of the alien flora in the Czech Republic and the historical dynamics of introductions at time scales of centuries; (ii) summarize available information on the patterns of invasion across landscapes and habitats in this country; and (iii) provide fact sheets of invasive neophytes in the Czech Republic including information on their impact and distribution. A comprehensive analysis of the structure and composition of the Czech alien flora, including other characteristics not reported here, such as taxonomic patterns, life histories, incidence in plant communities, habitat niche and pathways of introduction, can be found in the two editions of the *Catalogue of alien plants of the Czech Republic* (Pyšek et al. 2002b, 2012b), as well as other summary papers addressing plant invasions in this country (Chytrý et al. 2005, 2009b; Pyšek et al. 2012a).

8.2 Patterns in the Diversity of the Alien Flora

Based on a recent update, the alien flora recorded in the Czech Republic consists of 1454 taxa (species, subspecies, varieties, hybrids and cultivars) of vascular plants (see Pyšek et al. 2012b; their Appendix 2 for the list of taxa). This figure is slightly higher than 1434 alien taxa reported by Danihelka (2013), referred to in other chapters of this book, because the latter source did not count varieties and cultivars. The list of taxa (including varieties and cultivars) published decade earlier (Pyšek et al. 2002b) reported 1378 taxa. This increase by 76 taxa was not only due to newly arrived taxa recorded during the last decade but also due to a thorough exploration and taxonomic re-evaluation of the literature, herbaria and other sources (Pyšek et al. 2012b). The 1454 recorded taxa consist of 350 archaeophytes (plants introduced since the beginning of the Neolithic until the end of the Middle Ages; see Holub and Jirásek 1967; Pyšek et al. 2002b, 2004b for definitions) and 1104 neophytes (plants introduced in the Modern Period). Danihelka (2013) reports 345 taxa of archaeophytes and 1089 of neophytes. These two groups differ markedly in the numbers of taxa in particular categories along the introduction–naturalization–invasion continuum (INIC; following the concept of Richardson et al. 2000; Blackburn

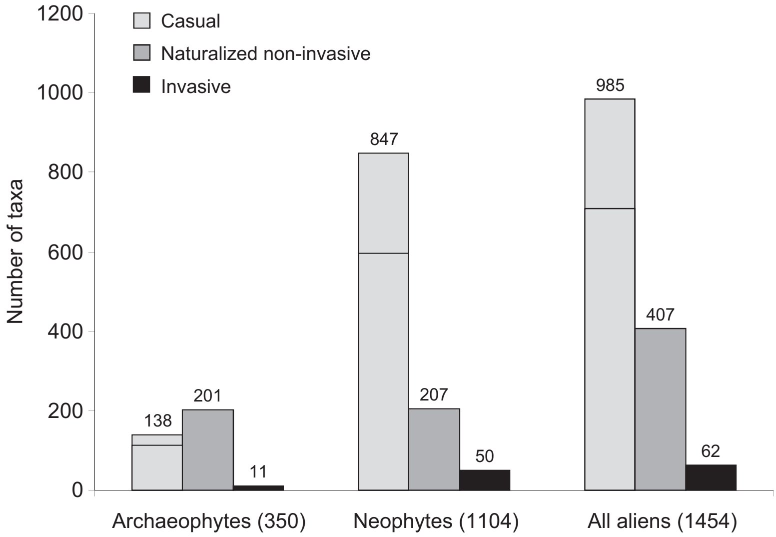

Fig. 8.1 Numbers of alien taxa in the Czech flora, shown separately for all aliens and the two subgroups distinguished with respect to their residence time, archaeophytes and neophytes. Taxa are classified according to the stage they have reached along the introduction–naturalization–invasion continuum (INIC), which describes how species proceed in the invasion process in terms of overcoming geographical, environmental and biotic barriers (Richardson et al. 2000; Blackburn et al. 2011). Within each group, taxa are divided into casual, naturalized but non-invasive, and invasive; the *upper part* of the bar representing casuals indicates the taxa that have vanished (taken from Pyšek et al. 2012a)

et al. 2011; Fig. 8.1; Table 8.1). While the numbers of naturalized but non-invasive archaeophytes and neophytes are similar (201 vs 207), there are more invasive taxa among the neophytes (50 vs 11). The markedly higher total taxonomic diversity of neophytes is due to the much higher number of casual taxa (847 vs 138; Fig. 8.1). Consequently, expressed in relative terms, the ratio of naturalized and casual species is the opposite in both groups: the majority of archaeophytes, but only a minority of neophytes are naturalized (archaeophytes: 60.6% vs 39.4% of casuals; neophytes: 23.3% vs 76.7%; Table 8.1). The percentage of naturalized taxa among all aliens increases, particularly for neophytes, if only plants assumed to be currently present are taken into account. If taxa that are considered to have vanished (i.e. recorded only once or a few times in the past and not observed for the last 25 years; see Pyšek et al. 2012b) are excluded, the opposite pattern remains the same (Table 8.1).

This difference in species richness of the two historical groups of alien species results partly from the fact that the number of archaeophytes is by definition (introduced up to 1500 A.D.) no longer increasing (Pyšek and Jarošík 2005). Even more relevant for species richness is that the archaeophytes we observe today are winners in the invasion process that has lasted for millennia and we have no information on the frequency of failures in the past (Pyšek et al. 2012b). In fact, current invasions

Table 8.1 Percentages of the different groups of alien taxa, based on their residence-time categories and invasion status, in the flora of the Czech Republic. Percentages of taxa in particular groups are shown for (a) the complete alien flora with all taxa ever recorded considered, and (b) taxa that may be assumed to occur at present (i.e. excluding the 277 taxa that have vanished from the complete flora). Note that 'invasive' is a subgroup of 'naturalized', therefore the total number of naturalized taxa is the sum of 'naturalized non-invasive', and 'invasive' (taken from Pyšek et al. 2012a)

	(a) All taxa (%)				(b) Currently present taxa (%)			
	Casual	Naturalized non-invasive	Invasive	Naturalized total	Casual	Naturalized non-invasive	Invasive	Naturalized total
Archaeo-phytes	39.4	57.4	3.2	60.6	34.6	62.0	3.4	65.4
Neophytes	76.7	18.7	4.6	23.3	69.9	24.1	6.0	30.1
All aliens	67.7	28.0	4.3	32.3	60.1	34.6	5.3	39.9

by neophytes tell a story of the archaeophytes' past, at least to some extent (Pyšek et al. 2011a). There is no reason to believe that invasions by archaeophytes were in principle different from modern invasions of neophytes in terms of introductions-and-failures, and booms-and-busts (sensu Williamson 1996; Blackburn et al. 2011) or that the dynamics were not similar to that recorded for neophytes. However, the intensity and frequency of these phenomena were probably smaller than today due to different pathways, lower propagule pressure, a smaller extent of human-induced disturbance and absence of invaders from overseas. The assumption of similar dynamics in the past is reflected in some archaeophytes being labelled as post-invasive; this category included species that now have a stable, or even decreasing population and do not invade new, modern habitat types, but their population dynamics and type of occurrence suggests that they might have been dominant in the vegetation in the past (see Pyšek et al. 2002b; Medvecká et al. 2012).

If archaeophytes and neophytes are merged into a single group of aliens, 67.7% of the taxa in the total Czech alien flora (985) are classified as casual and 32.3% (469) as naturalized. Separating the latter group according to the more advanced stages of the INIC indicates that 408 taxa (28.1% of the total alien flora) are currently naturalized but non-invasive, and 61 taxa (4.2%) are invasive (Fig. 8.1, see overview at the end of this chapter).

These figures correspond reasonably well with results from other European countries that have compiled complete checklists of their alien plants (see Lambdon et al. 2008; van Kleunen et al. 2015 for overviews). However, including a complete record of all the casual taxa on national or regional lists is still an exception rather than a rule. The comparability is also limited by the variation in the approaches to national checklists that may differ mainly in how criteria for naturalized or invasive taxa are applied, and whether only neophytes or also archaeophytes are included. Considering only neophytes (archaeophytes cannot be assessed because of missing information on unsuccessful casuals), the naturalization rate of 23.3% (i.e. the percentage of taxa in the total pool of introduced aliens that became naturalized) for the

Czech flora corresponds well with that recorded in other European countries such as 22.7% for Belgium (Verloove 2006), 20.5% for Hungary (Balogh et al. 2004), 25.4% for Austria (Essl and Rabitsch 2002) and 26.8% for Slovakia (Medvecká et al. 2012).

A comprehensive account of the alien flora in Slovakia (Medvecká et al. 2012) provides an excellent opportunity to compare the patterns of plant invasions in two neighbouring countries with shared culture and history, which were part of the same state until 1992, yet differ mainly in topographic heterogeneity and land use and hence in opportunities for invasion. A brief comparison of the two most recent catalogues (Medvecká et al. 2012; Pyšek et al. 2012b) indicates that overall, disproportionally fewer taxa are considered naturalized in the Czech Republic (28.1% of all aliens) than in Slovakia (39.1%), the difference being mainly due to archaeophytes (19.5% vs 39.4%). The proportions of casual, naturalized and invasive neophytes in the Czech Republic and Slovakia are very similar (75.7% vs 73.2%, 18.7% vs 22.8% and 4.5% vs 4.0%, respectively), indicating that in both checklists the transitions along the INIC are reasonably well captured.

8.3 Regional Distributions of Alien and Invasive Plants

Maps based on the percentage of alien species in 35 habitat types in the Czech Republic (Fig. 8.2), derived from quantitative data for ~19,000 vegetation plots (Chytrý et al. 2009b), show that both archaeophytes and neophytes are most common in lowland agricultural regions and urban areas, but sparsely represented in mountainous areas. At middle altitudes, agricultural areas are more invaded than forested areas. Outside agricultural and urban areas, high levels of invasion are found especially in lowland sandy areas and river corridors. The regional levels of invasion by neophytes follow a similar pattern whether expressed for all terrestrial habitats (Fig. 8.2a) or only natural and semi-natural habitats (Fig. 8.2b), but are generally lower in the latter, regardless of whether the relative numbers or covers of neophytes in vegetation plots are used (Chytrý et al. 2009b).

In addition to these maps that reflect the distribution of all aliens at the plant community scale, Pyšek et al. (2012a) published a map based on 26,462 floristic records of species that are considered invasive, which shows that the most invaded are the surroundings of big cities, floodplains of large rivers, post-mining disturbed landscapes in the northern part of the country, and warm lowlands in east-central Bohemia and southern Moravia (Fig. 8.2c). The hotspots of the occurrence of all aliens and currently spreading invaders are closely correlated. This confirms that the pattern recorded at the regional scale in the Czech Republic is similar to that previously highlighted at the continental scale in Europe by Chytrý et al. (2012); these authors report a strong positive relationship between the mean level of invasion by all aliens recorded in individual 50 km mapping grid cells and the percentage of the total number of the worst invasive plant species occurring in Europe present in these cells.

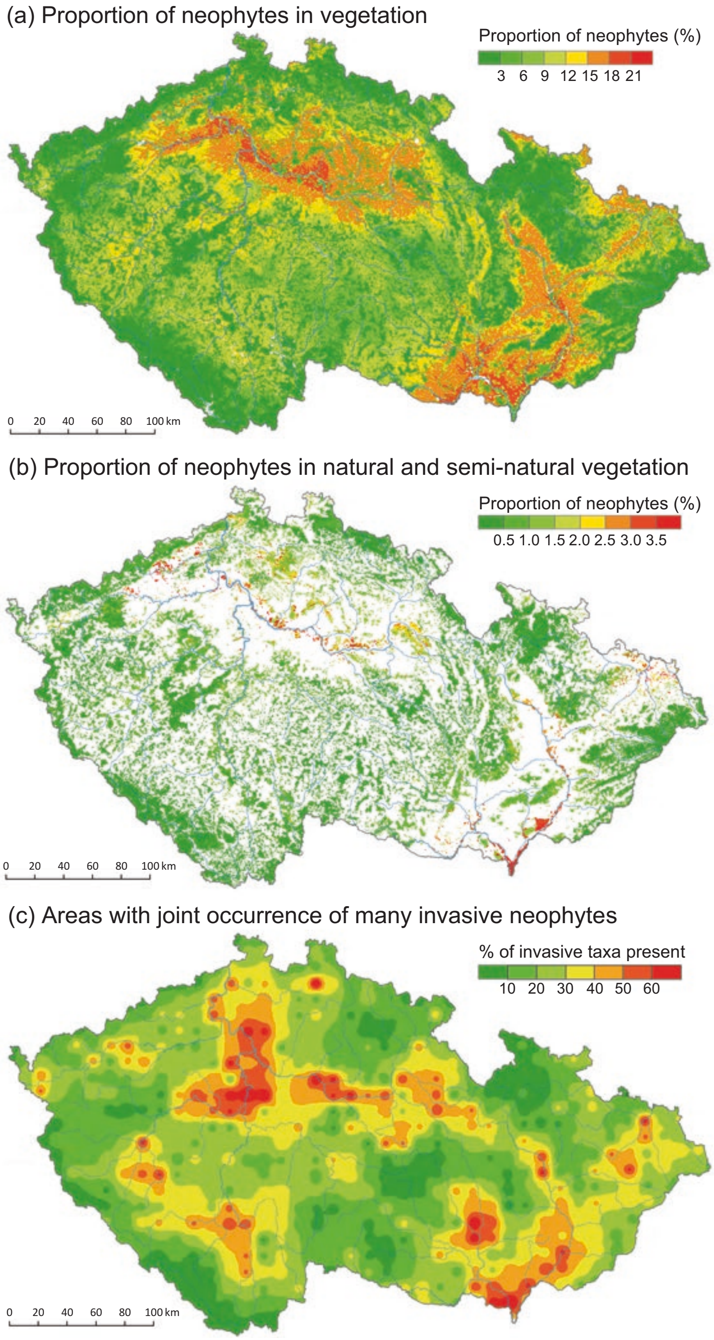

Fig. 8.2 Regional levels of invasion by neophytes in the Czech Republic. (**a**, **b**) Level of invasion by all neophytes regardless their invasion status in all vegetation types and separately in natural and semi-natural vegetation only. The values are percentage numbers of alien taxa among all taxa interpolated by generalized linear models for particular combinations of land-cover types and altitudes. (**c**) Levels of invasion based on the occurrence of 40 invasive neophytes. Percentages of invasive neophyte taxa present in grid cells of 6 × 10 min were interpolated using the inverse distance weighted interpolator applied to 12 neighbouring points (power parameter = 2). The location of cities and towns is indicated, with symbol size corresponding to their size (data were taken from Chytrý et al. 2009b: maps a, b, and Pyšek et al. 2012a: map c, and redrawn by O. Hájek)

8.4 Contribution of Aliens to Plant Diversity in the Czech Republic

The data available for both native (Danihelka et al. 2012) and alien flora (Pyšek et al. 2012b) in the Czech Republic make it possible to precisely estimate the enrichment of the national flora by alien species (Pyšek et al. 2003b). Considering all the taxa ever recorded in the national flora, i.e. including extinct natives and vanished aliens, gives totals of 1454 alien and 2945 native taxa and indicates that the former make up 33.1% (Fig. 8.3). Of this value, 8.0% are attributed to archaeophytes and 25.1% to neophytes. The total percentage in the flora is the figure usually reported in plant invasion literature. For European countries for which complete lists of casual species are available, the percentage of all aliens among the total flora ranges widely from rather low values in southern countries, such as 13% in Italy (Celesti-Grapow et al. 2009) and 12% in Spain (Sanz-Elorza 2004; cited by Celesti-Grapow et al. 2009), to those comparable with the Czech Republic recorded in Central European countries. These include e.g. Austria (27.0%, Essl and Rabitsch 2002), Hungary (26.6%, Balogh et al. 2004; number of native taxa from Winter et al. 2009) and Poland (27.3%, Tokarska-Guzik 2005; number of native taxa from Winter et al. 2009). The highest percentages of alien taxa are reported from the more northerly and westerly located regions in Europe, e.g. 35.3% in Estonia (Ööpik et al. 2008), 41.0% in Belgium (Verloove 2006; number of native taxa from Winter et al. 2009) and 53.4% in the UK (Lambdon et al. 2008), where higher percentages of aliens are partly attributed to the relatively species-poor native floras.

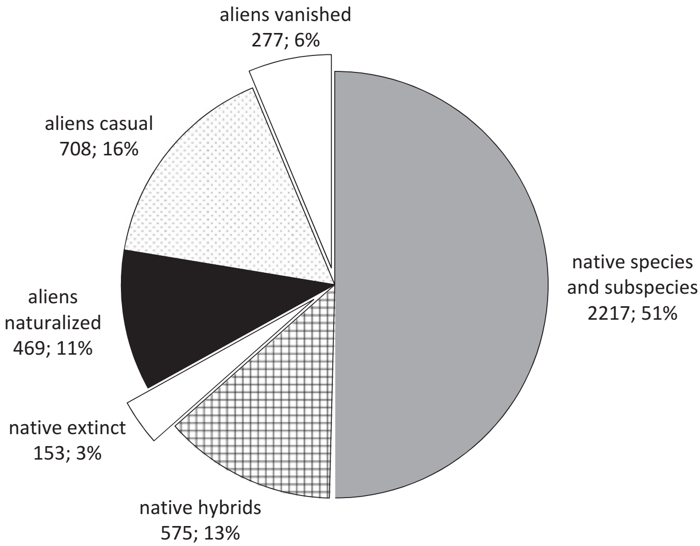

Fig. 8.3 Composition of the flora of the Czech Republic in terms of its native and alien status, categories of aliens, and historical and current presence of taxa in this country (data on the numbers in the native flora are from Kaplan 2012 and Danihelka et al. 2012, those on the alien flora from Pyšek et al. 2012b)

Similar estimates in the literature, however, depend on which taxa are included in the assessment of total plant diversity (e.g. subspecies, hybrids), how thorough and long-term is the recording of casual aliens, or whether or not archaeophytes are included, or distinguished at all (see Pyšek et al. 2002b; Williamson 2002; Lambdon et al. 2008 for discussion). This can be demonstrated using the Czech flora; excluding hybrids or taxa no longer present (extinct natives and vanished aliens) shifts the percentage of aliens to 36.4% or 29.7%, respectively. This is because hybridization among native species is more frequent (19.5%) than hybridization involving alien species (6.6%), hence the percentage increases. The opposite is true for extinct and vanished taxa, as extinctions account for 5.2% of the total native flora, but vanished taxa for 19.1% of the alien flora.

As the variation in the reported figures results not only from the composition of floras but also from other factors that introduce biases, invasion histories and regions can only be rigorously compared keeping these potentially biasing factors in mind. Again, neighbouring Slovakia is a country for which such a comparison is possible based on a recent checklist of alien species compiled using similar criteria. In this country alien taxa make up 21.5% of the total number ever recorded in its territory, of which 6.6% are attributed to archaeophytes and 14.9% to neophytes (Medvecká et al. 2012). The overall contribution of alien taxa is therefore markedly lower than the above estimate for the Czech Republic. Another possible comparison is of the taxa that are permanently present in the two countries, i.e. excluding extinct natives and including only naturalized alien taxa (Fig. 8.3). This measure yields a figure of 14.4% alien contribution to the permanent plant diversity in the Czech Republic, 6.5% and 7.9% attributed to archaeophytes and neophytes, respectively. For Slovakia, 373 alien taxa permanently present (206 naturalized archaeophytes and 167 naturalized neophytes; Medvecká et al. 2012) make up 10.3% of the contribution of alien taxa to the total plant diversity, a figure again by ~50% smaller than in the Czech Republic (Pyšek et al. 2012b).

Since both countries have similar sizes, climates, demographies and current macroeconomic parameters, i.e. factors that are known to determine alien species richness at a large scale (e.g. Vilà and Pujadas 2001; Pyšek et al. 2010), and a comparable intensity of floristic research, the explanation of these differences needs to be sought mainly in topography, land cover and land-use history. First, large parts of Slovakia consist of mountainous areas of the Western Carpathians, which are less invaded because of the general tendency of mountainous areas to be less invaded than areas at lower altitudes (Becker et al. 2005; Alexander et al. 2011; McDougall et al. 2011). Second, the Bohemian lands were industrialized already in the second half of the nineteenth century, while Slovakia was industrialized only after World War II. This earlier start of industry in the area that is now the Czech Republic undoubtedly intensified transportation and introduction of alien species.

8.5 Dynamics of Invasions Over Time

Pyšek et al. (2012b) assigned the year of the first record for 771 neophytes (i.e. 70% of the total number recorded) and showed that there has been a steady trend of approximately four new alien arrivals per year since the beginning of the nineteenth century without any distinct acceleration or deceleration. If the dynamics based on taxa with known year of the first record are projected to the total neophyte flora, the total number of estimated neophytes would reach 1264 by 2050 should the current trend persist (see Pyšek et al. 2012b for details). These trends suggest that the number of alien species recorded in the Czech Republic will be increasing at a similar rate in the near future, which corresponds to the trend reported for Europe (Lambdon et al. 2008; Hulme et al. 2009).

Displaying the dynamics of arrivals of alien neophytes separately by invasion status reveals that the rate of increase in the cumulative number of naturalized taxa somewhat decelerated in the second half of the twentieth century, while casuals exhibit an opposite trend, with a steep increase since the 1950s (Fig. 8.4). This acceleration is due to landscapes becoming more suitable for invasions after the dramatic changes in land-use that occurred after World War II (Müllerová et al. 2005; Williamson et al. 2005) but also, to some extent, reflects increased interest in alien plant research since the 1970s (Pyšek et al. 2002b, 2011b). However, the rapidly increasing numbers of neophytes are a warning because naturalized or even invasive species recruit from casuals with a delay of decades, due to lag phenomena and invasion debt (Kowarik 1995; Aikio et al. 2010; Essl et al. 2011).

The dynamics of arrivals also convincingly demonstrate the role of residence time in invasions; this is evident if naturalized species are categorized into naturalized but not (yet) invasive and invasive. Not only is the current distribution of neophytes positively related to minimum residence time, with those present for a longer time being more widely distributed or more abundant (Pyšek and Jarošík 2005;

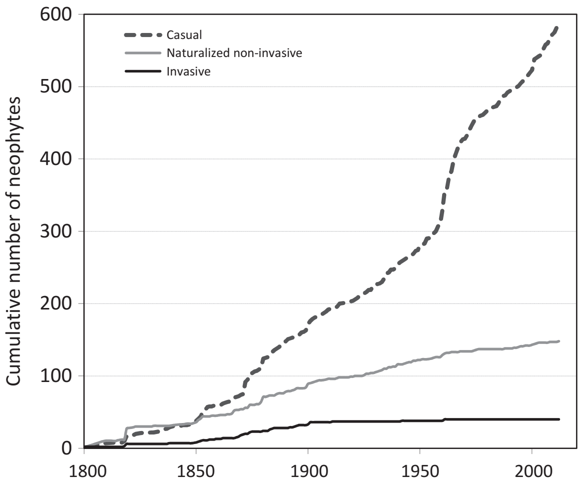

Fig. 8.4 Cumulative numbers of neophytes reported up to the given year shown separately for casual, naturalized non-invasive, and invasive species. Based on taxa for which the year of the first record is available (n = 771) (data from Pyšek et al. 2012a)

Rejmánek et al. 2005), but also residence time affects the invasion status of alien species. Of the neophytes currently classified as invasive in the Czech Republic (Pyšek et al. 2012b), 50% were introduced prior to 1872, i.e. earlier than naturalized but non-invasive (1886) and much earlier than casual neophytes (1956), or 90% of invasive, naturalized but non-invasive, and casual taxa were introduced before 1901, 1968 and 2001, respectively (Pyšek et al. 2012a).

The rates of species introductions differ not only based on invasion status but also on origin. An analysis of the source of alien taxa in the Czech flora by Pyšek et al. (2012b) revealed that the Mediterranean area (covering parts of Southern Europe, Northern Africa and Western Asia from Turkey and Israel to Afghanistan) is the main source not only of archaeophytes, 52.7% of which arrived from there, but also of neophytes (28.7%; Fig. 8.5). For neophytes, the other most important sources are extra-Mediterranean parts of Europe, contributing 19.9% of taxa, North America (16.7%) and extra-Mediterranean Asia (14.2%; Fig. 8.5). The dynamics of introduction of neophytes from these main donor regions indicate that the arrivals from the Mediterranean area and extra-Mediterranean Europe proceeded at the same speed until approximately the 1870s, after which introductions from the Mediterranean region became more frequent. In general, species native to more distant areas in Asia and North America arrived later (Fig. 8.6). In relative terms, however, the introductions of the extra-Mediterranean European species were the fastest: 50% of all currently known taxa from this region were in this country by 1895, compared to 1926 for the Mediterranean species, 1935 for North American species and 1958 for Asian species (Pyšek et al. 2012a).

The estimates of the long-term introduction dynamics of archaeophytes reveal that 35.2% of presently known taxa were introduced in the Neolithic/Chalcolithic period (5500–2200 years BC) and more than half (52.7%) are thought to have been

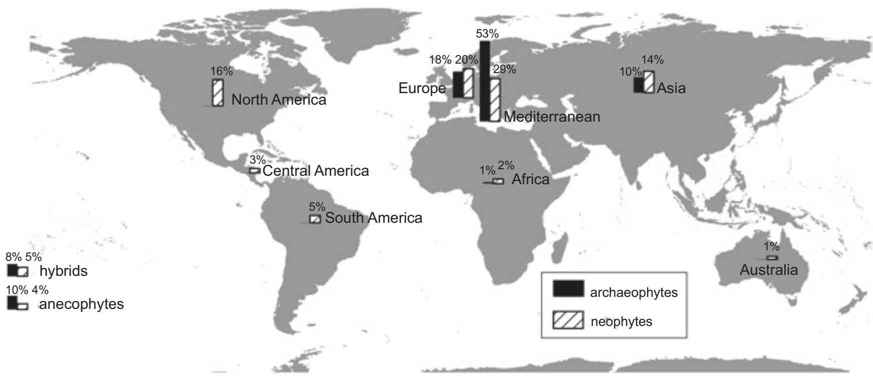

Fig. 8.5 Regions from which alien plants in the Czech Republic originated. The height of the bars reflects the percentage of the total number of taxa that are alien in the Czech Republic and native to particular regions (shown on top of the bar). The Mediterranean region includes parts of Southern Europe, Northern Africa and Western Asia from Turkey and Israel to Afghanistan. Europe, Asia and Africa refer to their parts other than those in the Mediterranean region in this delimitation (from Pyšek et al. 2012a)

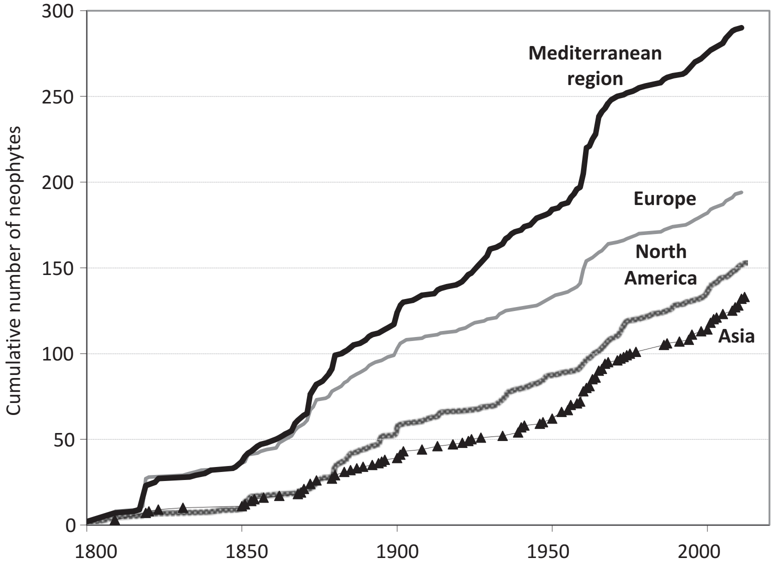

Fig. 8.6 Arrivals of alien neophytes in the Czech Republic from four main donor regions, shown as increase in the cumulative number of taxa originating from a given region. If a taxon originated from more than one region, only a fraction was attributed to each, e.g. ½ for a taxon originating from two regions (taken from Pyšek et al. 2012a)

present by the end of the Bronze Age, ~750 years BC (Pyšek et al. 2003b). The temporal dynamics of archaeophytes in the Czech flora also reflects a clear effect of residence time, detectable after centuries to millenia since the start of their invasions. Taxa that are most widespread at present were introduced earlier than those currently less widely distributed (Pyšek and Jarošík 2005).

8.6 Invasive Taxa in the Czech Republic

In the checklist of alien flora in the Czech Republic 61 taxa are labelled as invasive (Pyšek et al. 2012b). This work followed the definition of an 'invasive species' as one that forms self-replacing populations over many life cycles, produces reproductive offspring, often in very large numbers at considerable distances from the parent population or site of introduction, and has the potential to spread over long distances (Richardson et al. 2000; Pyšek et al. 2004b; Blackburn et al. 2011). In addition to this definition, Pyšek et al. (2012b) introduced the metapopulation criterion to separate invasive species from naturalized, in order to account for the historical population dynamics of the respective taxa, and classified the invasion status based on the population history viewed from the current perspective, i.e. the state in which the populations of a given species exist at present. Therefore, some taxa previously considered invasive are now classified as naturalized, reflecting the 'boom-and-bust phenomenon' (sensu Williamson 1996; Strayer et al. 2017). Another principle adopted was that of the highest stage achieved at the population level, reflecting that individual populations of an alien species may occur in a region at different stages of the INIC (e.g. Essl et al. 2009; Richardson and Pyšek 2012). Therefore, if some populations of a species reach the invasion stage, the species is classified as invasive (see Pyšek et al. 2012b for details on the approach).

Among the taxa currently considered as invasive there are 11 archaeophytes (*Angelica archangelica* subsp. *archangelica*, *Arrhenatherum elatius*, *Atriplex sagittata*, *Cirsium arvense*, *Conium maculatum*, *Digitaria ischaemum*, *Echinochloa crus-galli*, *Eragrostis minor*, *Portulaca oleracea* subsp. *oleracea*, *Prunus cerasifera* and *Stellaria pallida*) and 50 neophytes. Invasive neophytes occur in a wide range of habitats (Figs. 8.7 and 8.8) and are addressed in detail in Sect. 8.6, where the information on their ecology, invasion history, current distribution, trends and impact is summarized.

8.7 Patterns of Plant Invasions Across Landscapes and Habitats in the Czech Republic

Important insights into the patterns of invasion of different habitats and plant communities are gained by combining the knowledge on alien flora with approaches of vegetation ecology (Pyšek and Chytrý 2014). Plant invasion patterns across landscapes, habitats or vegetation types can be quantified in terms of the level of invasion, measured as the number of alien species, or the percentage of aliens to all species, per unit area. The level of invasion is influenced by habitat invasibility (i.e. its vulnerability to invasion; Rejmánek 1989) and propagule pressure of alien species (Lonsdale 1999; Chytrý et al. 2008a). The latter can be understood either as the number of propagules arriving at a given location (propagule pressure sensu stricto) or the number of species (colonization pressure; Lockwood et al. 2009). Technically, habitat invasibility can be quantified by separating the effect of propagule pressure from the level of invasion measured at particular sites (Lonsdale 1999; Chytrý et al. 2008a). However, it is difficult to obtain an exact measurement of propagule pressure. Therefore studies of invasibility are limited to the use of proxy variables such as the type of land use in the surroundings of the sites sampled. In the Czech Republic such a study indicated a rather strong relationship between the level of invasion and habitat invasibility: generally, invasible habitats tend to be invaded to a high level and vice versa, although there are exceptions (Chytrý et al. 2008a). Unfortunately, for habitats restricted to areas with limited propagule pressure, it cannot be decided by observational studies whether they are invasible or resistant to invasions.

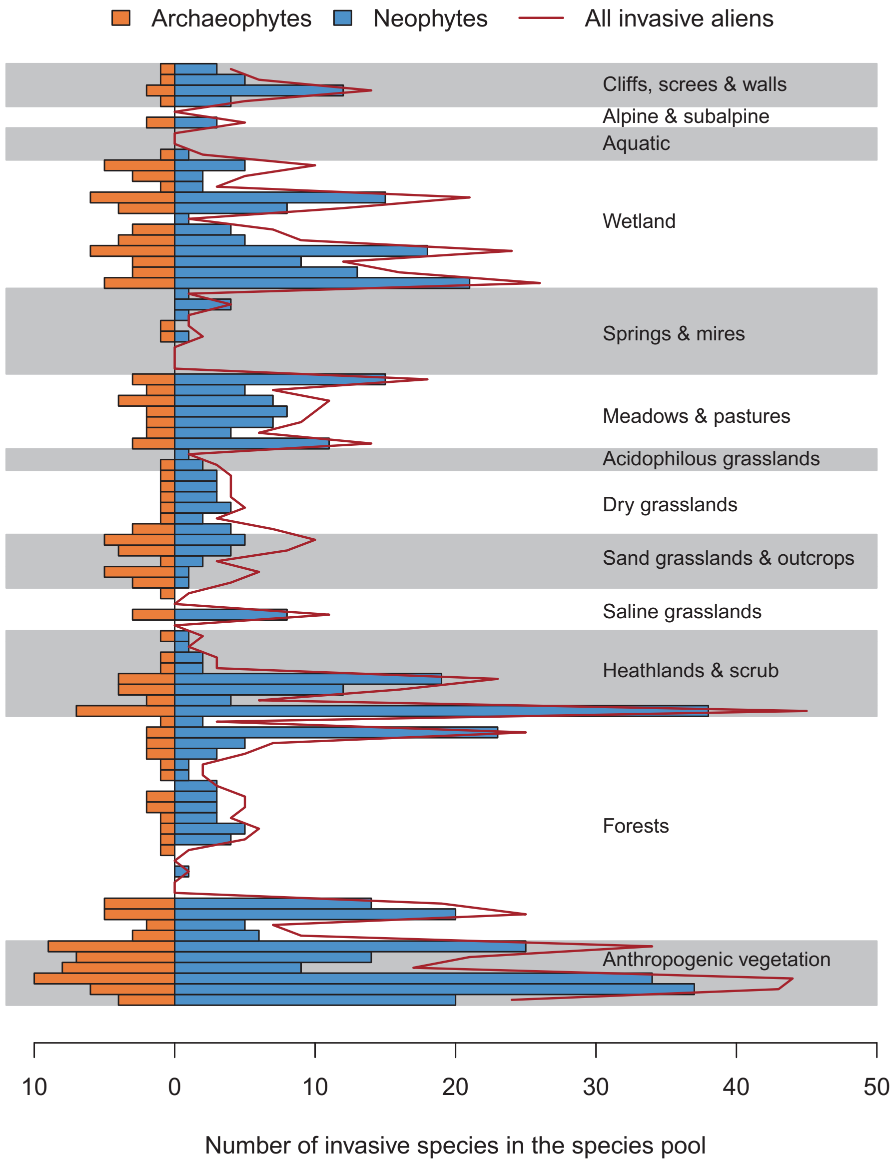

Fig. 8.7 Numbers of invasive archaeophytes (n = 11) and invasive neophytes (n = 50) in different habitats (n = 88) in the Czech Republic, based on taxa listed in this chapter (data from Sádlo et al. 2007 and Pyšek et al. 2012a)

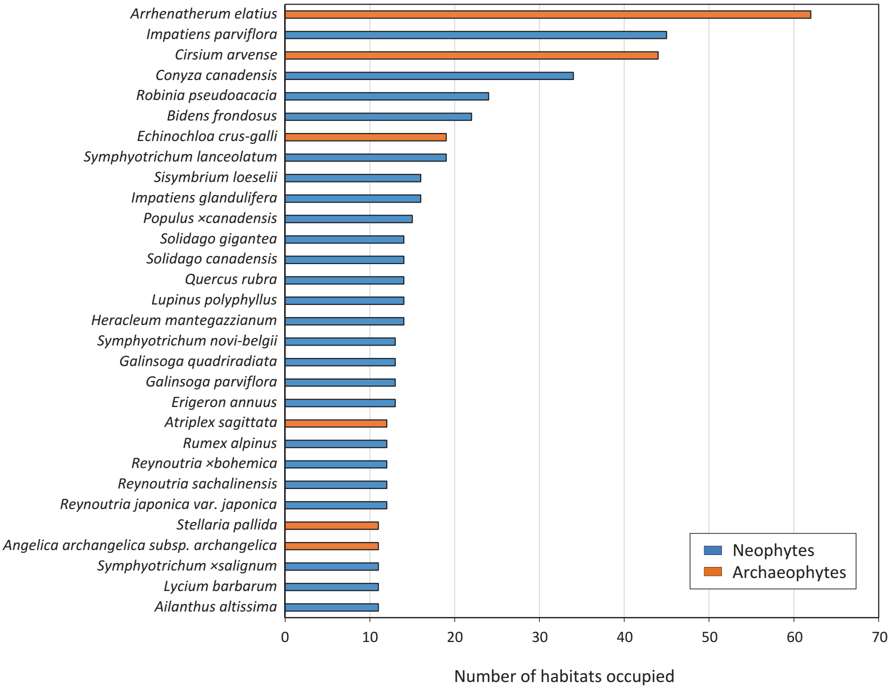

Fig. 8.8 Numbers of habitats invaded by archaeophytes and neophytes that are considered invasive in the Czech Republic. The classification of habitats follows that used in Sádlo et al. (2007; n = 88). Only taxa occurring in more than 10 habitats are shown (based on data in Pyšek et al. 2012a)

8.7.1 Level of Invasion and Altitude

A decrease in the level of plant invasions at high altitudes is a globally consistent pattern. Generally, alien floras in mountain areas are subsets of alien floras in the surrounding lowlands, the former consisting of species with the broadest climatic tolerance, which continuously spread from the foothill regions (Alexander et al. 2011; McDougall et al. 2011). For example, Pyšek et al. (2011c) demonstrated that neophytes in the Czech Republic were most often introduced into areas located at 250–400 m a.s.l., from where they subsequently spread to higher altitudes. This is supported by the fact that species with earlier first records (thus, presumably introduced earlier) tend to have a broader altitudinal range. The altitudinal pattern of invasion can be explained by the low-altitude filter effect, limited landscape connectivity in the mountains and gene flow from lowland populations to peripheral populations at high altitudes, which would hinder the development of adaptations to high-altitude environments (Becker et al. 2005). In addition, the available total land area decreases with altitude, which may further contribute to lower levels of invasion, e.g. by reducing the diversity and extent of suitable habitats and thereby reducing the frequency of invasion foci for further spread.

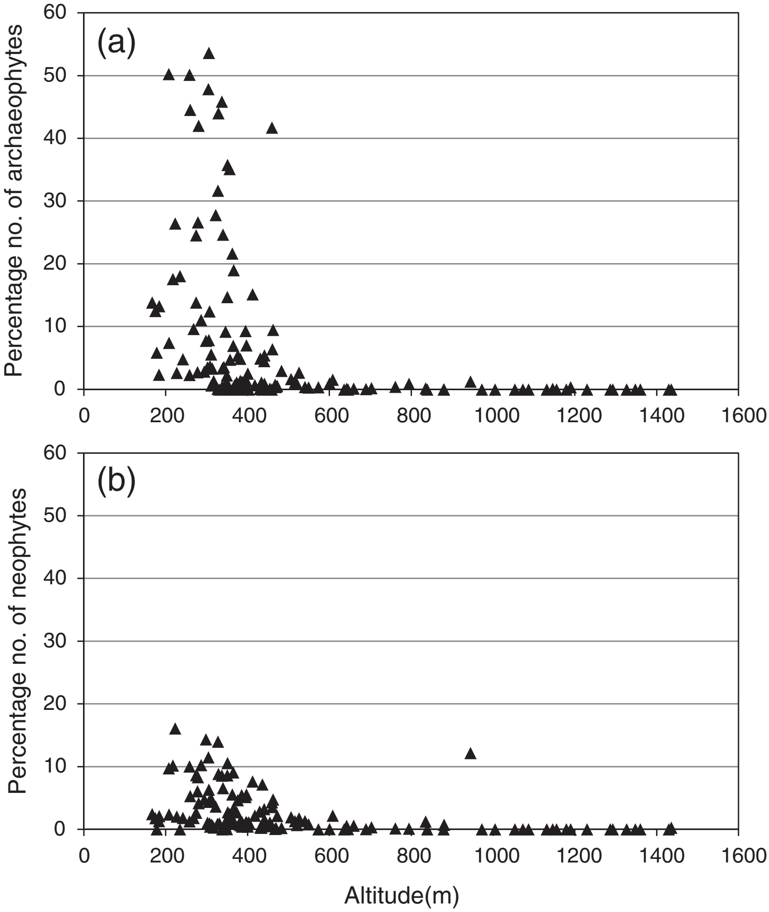

Fig. 8.9 Percentage of (**a**) archaeophytes and (**b**) neophytes, relative to the number of all species recorded in vegetation plots, plotted against altitude. Values are means for phytosociological alliances recognized in the Czech Republic. Percentages of alien species are from Fig. 8.10 and altitudes were taken from the vegetation plots used as the data source for that figure. Note that the altitudinal range in the Czech Republic is 115–1603 m. The outlier on the right side of the lower graph is the alliance *Rumicion alpini* (taken from Pyšek et al. 2012a)

Although the altitudinal range in the Czech Republic (115–1603 m) is not very broad, the decrease in the level of invasion with altitude is distinct and well documented (Fig. 8.9). It has been demonstrated in comparative studies of landscape segments, e.g. using a transect through a landscape in southern Bohemia that also included an altitudinal gradient (Mihulka 1998), Czech nature reserves (Pyšek et al. 2002a) and grid cells of flora mapping in the White Carpathian Mts in south-eastern Moravia (Otýpková et al. 2011). A possible explanation of this pattern might be that higher altitudes generally include smaller areas of invasible habitats. However, analyses performed within individual habitats, using vegetation plots, also detect a decreasing trend in the level of invasion with altitude, e.g. for weed communities on arable land (Lososová et al. 2004; Pyšek et al. 2005), ruderal vegetation (Simonová and Lososová 2008) and most other habitat types in the Czech Republic except those with narrow altitudinal ranges (Chytrý et al. 2009b). In most contexts, neophytes respond to altitude more strongly than archaeophytes, the former being more distinctly concentrated in the lowlands.

8.7.2 Level of Invasion Across Habitats

Previous research (Chytrý et al. 2005) identified arable land and anthropogenic ruderal vegetation as the habitat types harbouring the highest percentages of both archaeophytes and neophytes. Deciduous broad-leaved forest plantations (of e.g. *Populus* ×*canadensis*, *Quercus rubra* and *Robinia pseudoacacia*) are also frequently invaded, especially by neophytes. These habitats not only have the largest percentage of alien species in small plots but also the largest regional pools of alien species, i.e. species occurring in the Czech Republic and adapted to these habitats (Sádlo et al. 2007). Levels of invasion for more finely defined vegetation types, basically at the level of phytosociological alliances, are quantified for ruderal vegetation (Simonová and Lososová 2008), weed vegetation (Lososová and Grulich 2009) and forests (Chytrý et al. 2009b).

An analysis of the habitat species pools as defined in Sádlo et al. (2007) revealed that there is a strong phylogenetic signal in the invasion process for both the invaders and the invaded vegetation (Lososová et al. 2015). The numbers of alien species in particular habitat types in the Czech Republic are related both to the phylogenetic structure of plant communities occurring in these habitat types and the phylogenetic position of the invading alien species. Frequently disturbed herbaceous vegetation with strong phylogenetic clustering is more invaded than other vegetation types, possibly due to disturbance acting as an environmental filter. However, alien species not only invade more massively those community types that are phylogenetically clustered but also increase the post-invasion degree of phylogenetic clustering of those communities as they tend to be from the same lineages as native species. By this process, invaded community types become even more phylogenetically clustered. These findings support the hypothesis that the relatedness of invaders to native species promotes invasion because they share adaptations to the same environments. However, such trends are not detected for phylogenetically more diverse and less invaded communities, such as forests (Lososová et al. 2015).

Here we provide a new quantification of the level of invasion for plant communities in the Czech Republic at the level of phytosociological alliances, following the classification system accepted in the monograph Vegetation of the Czech Republic (Chytrý 2007–2013) and the most recent edition of the Catalogue of alien species of the Czech Republic (Pyšek et al. 2012b; Fig. 8.10). It is based on a stratified-random selection of 20,830 vegetation-plot records (relevés) from the Czech National Phytosociological Database (Chytrý and Rafajová 2003) and prepared using the

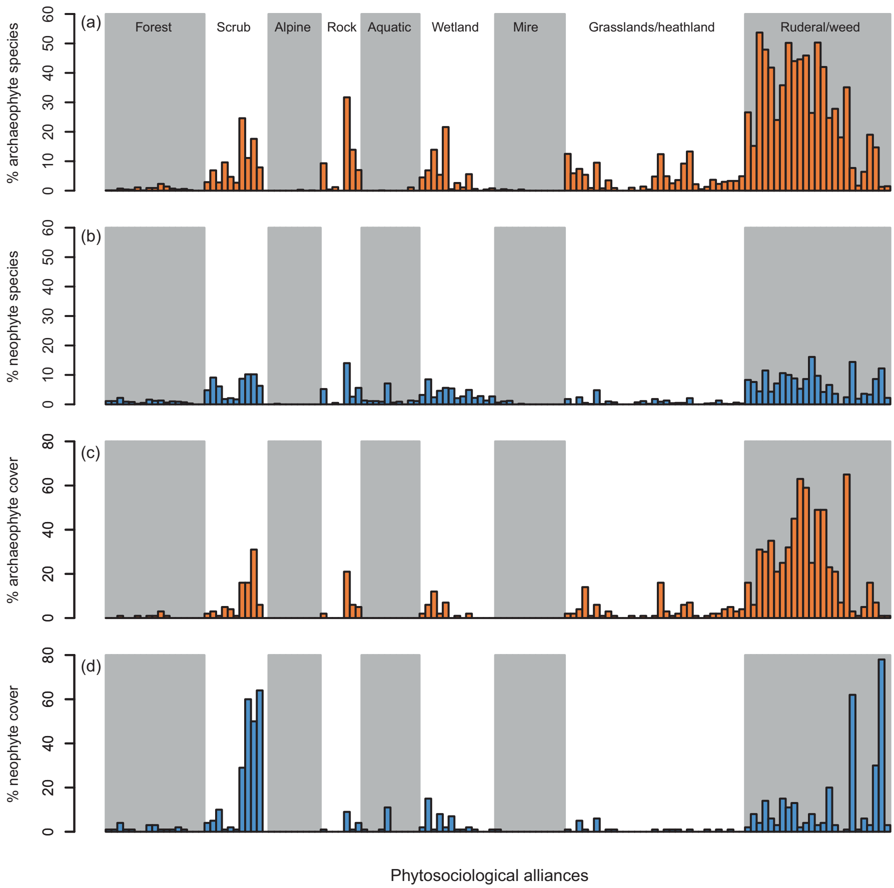

Fig. 8.10 Mean percentages and mean summed covers of archaeophytes and neophytes, relative to all species recorded in vegetation plots, calculated for phytosociological alliances in the Czech Republic. The graphs are based on 20,830 plots of all vegetation types from the Czech National Phytosociological Database, excluding those made before 1980 and in plot sizes deviating from standards for a given vegetation type. The plots were classified to phytosociological associations using an expert system for automated classification developed as a part of the project Vegetation of the Czech Republic (see Chytrý 2007–2013 for details; based on data presented in Pyšek et al. 2012a: Table 3; see this paper for more details)

same methods as the previous quantification (Chytrý et al. 2005), but it uses more finely defined vegetation units and newer plots sampled since 1980. Compared to previous analyses (Chytrý et al. 2005, 2009b; Simonová and Lososová 2008; Lososová and Grulich 2009), the new results report slightly lower numbers of archaeophytes and higher numbers of native species in some cases, which is mainly due to reclassification of 41 species considered as archaeophytes by Pyšek et al. (2002b) to native in Pyšek et al. (2012b). However, the general trend remains the same as reported in previous studies.

The most invaded habitats and vegetation types of the Czech Republic, in terms of the percentages of alien species they harbour, are generally either disturbed or have fluctuating inputs of resources, especially nutrients and in some cases water or light. A comparison with corresponding habitats in other European countries shows that such habitats are also among the most invaded elsewhere, although the composition of alien floras is very different among European regions (Chytrý et al. 2008b). In contrast, habitats with limited fluctuation of resource availability such as dry, wet and saline grasslands, base-rich fens and broad-leaved deciduous woodlands appear to be rather resistant to invasion, although they can be invaded to some extent in areas subject to high propagule pressure (Chytrý et al. 2008a). Similar patterns are recorded in comparative studies of invasions of habitats in the Czech Republic and eastern North America (Kalusová et al. 2014, 2015). This supports the theory of fluctuating resource availability as an important driver of community invasibility (Davis et al. 2000).

Sites, areas or habitats with a high percentage of neophytes usually also have a high percentage of archaeophytes, as shown by the analyses of plant species lists from Czech nature reserves (Pyšek et al. 2002a), grid mapping of flora at a landscape scale (Otýpková et al. 2011) and vegetation plots from the whole Czech Republic (Chytrý et al. 2005). However, there is some variation around this general trend. Archaeophytes are often more numerous in treeless vegetation on dry to mesic soils, while neophytes are more common in disturbed woody vegetation, wetlands or aquatic habitats (Chytrý et al. 2005, 2008b). On arable land in the Czech Republic there are proportionally more archaeophytes in areas with less precipitation and on drier soils, such as chernozem or rendzina, although neophytes are also common there (Pyšek et al. 2005). This pattern obviously results from the fact that most archaeophytes originate from dry areas in the Mediterranean region, including the Middle East (Pyšek et al. 2012b), and are therefore preadapted to dry open habitats (di Castri 1989; Pyšek and Jarošík 2005).

The incidence of alien species (their number, percentage of all species, or cover) is generally higher in early successional stages and decreases with successional age (Rejmánek 1989). This pattern is recorded in different successional seres in the Czech Republic, for example in abandoned fields in dry areas of the Bohemian Karst (central Bohemia; Rejmánek 1989) or southern Moravia (Sojneková and Chytrý 2015), southern Bohemian bogs disturbed by peat extraction (Bastl et al. 1997), disused sand or gravel pits across the country (Bastl et al. 1997; Řehounková and Prach 2008) and some other early-successional habitats (Prach et al. 2008). However, the course of succession can be changed if strongly competitive alien species occur near the successional site. For example, Řehounková and Prach (2008) report that the occurrence of mature *Robinia pseudoacacia* trees within 100 m of a disused sand or gravel pit resulted in the formation of *Robinia* groves during spontaneous succession in the pit.

The most invaded areas in the Czech Republic are agricultural landscapes with predominantly arable fields, cities and villages, and forest plantations in the lowlands. River floodplains, especially in the lowlands, are also strongly invaded (Vymyslický 2001; Matějček 2008; Kalusová 2009; Kalníková 2012; Šenová and Matějček 2013). Mid- and high-altitude landscapes are less invaded, especially if they are forested (Chytrý et al. 2009b). In a European context, the Czech Republic belongs to a strongly invaded area of Western and Central Europe, which is characterized by on average higher levels of invasion than the boreal zone of Northern Europe or the Mediterranean and sub-Mediterranean zones in Southern Europe (Chytrý et al. 2009a).

Habitat type is the most important determinant of the level of plant invasion in the Czech Republic, followed by altitude (and associated effects of climate) and variation in propagule pressure (Chytrý et al. 2008a). However, the patterns in the invasion of habitats also depend on the original habitats in which the introduced species occur in their native ranges. Hejda et al. (2009b) show that most neophytes occurring in the Czech Republic originate from various types of ruderal vegetation, dry grasslands, broad-leaved deciduous woodlands, moist and wet grasslands including tall-herbaceous vegetation, cliffs and rock outcrops, arable land and mesic grasslands. Especially those neophytes that originate from riverine habitats, eroded slopes and avalanche tracks are the most likely to become invasive, once they are introduced into Central Europe (Hejda et al. 2009b).

The Czech flora was recently used for investigating the biogeographic aspects of habitat invasions, in particular the role of habitats as sources of species that are invasive on other continents. Dostál et al. (2013) demonstrated that the native species in the Czech flora that originate from more productive habitats or have a broader niche are more successful invaders elsewhere in the world. Outside their native ranges, Central European species invade habitats that are similar to their habitats in Central Europe (Kalusová et al. 2013, 2014). European habitats that are important donors of alien species for other continents experience the highest levels of invasion by alien species from other regions (reciprocal species-pool effect; Kalusová et al. 2014). Hejda et al. (2015), in their global analysis of habitat affinities of alien species in their native ranges, took into account the direction of invasion and showed that European grassland species are much more successful as world-wide invaders than are grassland species from other continents in invading Europe. Conversely, New World wetland species invading Europe are more successful than *vice versa*. These authors also showed that successful invaders are adapted to a broad spectrum of successional phases in their native range, ranging from grasslands to forests. A comparison of the level of invasion in corresponding habitats in the Czech Republic and eastern North America demonstrates that although the pattern in the level of invasion is very similar in both areas, North American habitats are more invaded at the habitat scale and to some extent also at the plot scale (Kalusová et al. 2015). Interestingly, the direction of invasion between continents also affects how strong the impact on native biota will be. Hejda et al. (2017) show that European invaders have more profound impacts in North America than North American invaders in Europe.

8.7.3 Changes in the Level of Invasion over Time

There are distinct temporal trends in the percentages of alien species in Czech vegetation, but these trends are opposite in archaeophytes compared to neophytes. Lososová and Simonová (2008) compared data on the ruderal and weed flora in Moravia between the early twentieth century and the turn of the twenty-first century, showing a decrease in archaeophytes and increase in neophytes. The same trends for these two groups of aliens were confirmed by analyses of data spanning a few decades in the second half of the twentieth century, e.g. for arable weed vegetation throughout the Czech Republic (Lososová et al. 2004; Pyšek et al. 2005), ruderal vegetation in the city of Plzeň (Pyšek et al. 2004a; here the increase in neophytes was non-significant) and road side vegetation in the Orlické hory Mts and their foothills (Dostálek et al. 2016). These trends will probably continue in the future, although the future spread of alien species will mainly depend on changing land use. European scenarios of future changes in the level of invasion project smaller increases in Central Europe than in North-western Europe (Chytrý et al. 2012). The mean level of invasion across landscapes may even decrease slightly in regions where large areas of arable land are abandoned, leading to a subsequent decrease in alien species during succession.

8.8 Impact, Management and Legislation

In recent years, an important new avenue of research into biological invasions has focused on describing, classifying and categorizing impacts of alien biota (see Vilà et al. 2010, 2011; Pyšek et al. 2012c for the most comprehensive global accounts on plants). A system for classifying alien species according to the magnitude of their environmental impacts, based on the mechanisms of impact, has been proposed (EICAT; Blackburn et al. 2014) and recently adopted as an official tool by IUCN, similar to their Red List scheme. A GISS (Generic Impact Scoring System) scheme based on semi-quantitative scores assigned to impact categories allows not only environmental but also socioeconomic impacts to be assessed (Kumschick et al. 2015; Nentwig et al. 2016). Using GISS, Rumlerová et al. (2016) assessed the impacts of 128 alien plant species in Europe, of which 86 are known to occur in the Czech Republic (Table 8.2).

Among the 20 taxa invasive in Europe with the highest scores of summary impact, there are 11 that are also invasive in the Czech Republic; these plants pose the greatest threat to the natural environment. The scores for environmental impacts are generally higher than those for socioeconomic impacts, reflecting that to some extent, rigorous data on the latter are still rather scarce. Table 8.2 shows several taxa with potentially great impacts, both environmental and socioeconomic, that are however not realized due to their casual occurrence in the Czech Republic, mostly due to the unsuitable climate.

The Czech Republic is still lacking a comprehensive assessment of the ecological and economic impacts of its alien plants. Studies rigorously testing ecological

Table 8.2 Twenty alien species occurring in the Czech Republic that rank highest based on their potential impacts in Europe. Taken from Rumlerová et al. (2016) who assigned scores for all 86 species assessed for Europe and occurring in Czech Republic. Summary scores across six types of environmental and socioeconomic impacts are used as the measure, with each type scored 1–5 (see Nentwig et al. 2016 for details). Environmental impacts include (1) direct impact on plants; (2) impact on animals; (3) indirect impacts on other species; (4) transmission of diseases and parasites; (5) impact through hybridization; and (6) impact on ecosystems. Socioeconomic impacts concern (1) agricultural production; (2) animal production; (3) forestry; (4) human infrastructure; (5) human health; and (6) human social life. Species are ranked according to the sum of their impacts. Their status in the Czech Republic is given in brackets (based on Pyšek et al. 2012b); invasive species are in *bold*

Species	Environmental	Socioeconomic	Sum
Eichhornia crassipes (cas)	16	13	29
Elodea canadensis (nat)	15	8	23
***Reynoutria japonica* (inv)**	12	9	21
***Heracleum mantegazzianum* (inv)**	13	8	21
***Robinia pseudoacacia* (inv)**	11	9	20
***Solidago canadensis* (inv)**	14	4	18
***Ambrosia artemisiifolia* (inv)**	9	8	17
***Prunus serotina* (inv)**	12	5	17
Ambrosia trifida (cas)	10	6	16
Arctotheca calendula (cas)	7	9	16
***Conyza canadensis* (inv)**	8	8	16
Elaeagnus angustifolia (cas)	11	5	16
***Ailanthus altissima* (inv)**	11	4	15
Rosa rugosa (cas)	12	3	15
Elodea nuttallii (cas)	8	6	14
***Impatiens glandulifera* (inv)**	10	4	14
***Lupinus polyphyllus* (inv)**	11	3	14
Oxalis pes-caprae (cas)	9	5	14
***Bidens frondosus* (inv)**	7	6	13
Datura stramonium (nat)	5	8	13

impacts under local conditions are few (Hejda et al. 2009a; Horáčková et al. 2014; Jandová et al. 2014) and similar to the GISS-based one presented above (Table 8.2), but are a good beginning. A thorough nationwide assessment of ecological and economic impacts of alien plants in the Czech Republic is one of the most important future tasks for both researchers and practitioners. A more systematic scoring of impacts is currently underway and urgently needed in order to provide state authorities and responsible management bodies with a knowledge base that can be used to effectively mitigate the problem of plant invasion at the national level. In terms of costs incurred by invasions of alien species, information is only available for small (mostly protected) areas and specific eradication measures and management efforts. These data relate to individual invasive species, such as *Pinus strobus* in the Bohemian Switzerland National Park, for which a management cost of almost 1.5 million EUR was estimated for the period of 2000–2016 (including costs associated with management of the locally non-native *Larix decidua*; see Chap. 9, Sect. 9.7.3, this book). Another example is the management of six major invasive taxa subjected to control in the Bohemian Paradise (Český ráj) Landscape Protected Area in 2003, where the cost reached 450,000 CZK (Křivánek 2006). A study of *Heracleum mantegazzianum* estimates that the total annual economic impact of this species in the Czech Republic is 2.5 million CZK (Linc 2012).

The adoption by the European Union of Regulation 1143/2014 on invasive alien species (IAS; European Union 2014) is currently the most important policy instrument related to biological invasions in Europe. The backbone of this regulation is a list of species alien to the EU and identified as invasive based on a detailed evidence-based risk assessment. The species listed are banned from being imported, traded, possessed, bred, transported, used and released into the environment (Genovesi et al. 2014). In February 2016, the European Commission adopted a first list of IAS of Union concern consisting of 37 species, selected from the 65 species for which fully compliant risk assessments were available (Roy et al. 2015). Among the 13 plant species on this list, only *Eichhornia crassipes* and *Heracleum persicum* are recorded from the Czech Republic, but both only as casuals. Although the selection of species for this list has been criticized as being influenced by political interests (Pergl et al. 2016b), this Regulation is an important milestone for an invasive species policy in Europe. In the Czech Republic, the key legislative instrument for biodiversity conservation is Act no. 114/1992, which restricts deliberate introductions of non-native species into the wild. Another important legislative tool is Act no. 326/2004 on phytosanitary care, which focuses on weeds and sets the obligation to minimize the impact of alien species on nature. There are also other acts related to particular sectors such as forestry (no. 289/1995), agriculture (no. 334/1992), water resources (no. 254/2001) or genetically modified organisms (no. 78/2004) that concern alien species and their introductions (Šíma 2008).

8.9 National Black List of Invasive Plants

Intensive research focused on the categorization of alien plant species in the Czech Republic based on their invasiveness and the recent beginning of systematic scoring of impacts mentioned above (Kumschick et al. 2015; Rumlerová et al. 2016), yielded robust knowledge that allowed the preparation of the first national Black, Grey and Watch Lists of invasive biota (Pergl et al. 2016d). The Black List classifies species to three categories defined on the basis of the magnitude of the impact and pathway of introduction that affect the opportunities for action; for each category management measures are recommended that can be used by land managers, policy makers and other stakeholders. Species with a low impact, but for which some level of management and regulation is desirable, are included on the Grey List. Potentially dangerous species occurring in European countries with similar climates, as well as those introduced in the past but with currently no known wild populations in the Czech Republic, are included on the Watch List. In total, there are 78 plant species on the Black List, 47 on the Grey List, and 25 on the Watch List (Table 8.3).

Table 8.3 Categories of the Black List with indications of recommended management, handling restrictions, examples of species and criteria for classifying that are derived from environmental and socio economic impact, population status and distribution of the target species (based on Pergl et al. 2016b)

	Grouping criteria	Population status and distribution	Recommended local management	Handling and release restrictions	No.	Species
BL1	High environmental and socio economic impact	Abundant, occurring in a wide range of habitats throughout the country	Complete eradication, disposal of abandoned plantations	No release, application of trade regulations	2	*Ambrosia artemisiifolia*, *Heracleum mantegazzianum*
BL2	Moderate to massive environmental impact; depending on human actions that promote species spread	Often found as remnants of planting in gardens and plantations, usually with a wide distribution	Stratified approach; promoting alternative native species; permit only in areas with low conservation value; disposal of abandoned plantations	No release, legislative regulations of trade, handling, and planting	49	*Azolla filiculoides*, *Acer negundo*, *Ailanthus altissima*, *Allium paradoxum*, *Amorpha fruticosa*, *Arrhenatherum elatius*, *Asclepias syriaca*, *Beta vulgaris Altissima* Group, *Buddleja davidii*, *Colutea arborescens*, *Cornus alba*, *C. sericea*, *Cytisus scoparius*, *Echinocystis lobata*, *Echinops exaltatus*, *E. sphaerocephalus*, *Fallopia aubertii*, *Fraxinus pennsylvanica*, *Galega officinalis*, *Galeobdolon argentatum*, *Helianthus ×laetiflorus*, *H. pauciflorus*, *H. tuberosus*, *Impatiens glandulifera*, *Laburnum anagyroides*, *Lupinus polyphyllus*, *Lycium barbarum*, *Parthenocissus inserta*, *P. quinquefolia*, *Physocarpus opulifolius*, *Phytolacca acinosa*, *Pinus nigra*, *P. strobus*, *Populus ×canadensis*, *P. balsamifera*, *Prunus cerasifera*, *P. serotina*, *Pyracantha coccinea*, *Quercus rubra*, *Reynoutria ×bohemica*, *R. japonica*, *R. sachalinensis*, *Rhus typhina*, *Robinia pseudoacacia*, *Rudbeckia laciniata*, *Solidago canadensis*, *S. gigantea*, *Symphoricarpos albus*, *Symphyotrichum novi-belgii*, *Telekia speciosa*
BL3	Moderate to massive environmental impact; results of unintentional introductions	Usually with a wide distribution resulting from spontaneous spread	Stratified approach; due to spontaneous spread there is no need to tolerate in any area	No release	27	*Abutilon theophrasti*, *Alopecurus myosuroides*, *Amaranthus albus*, *A. powellii*, *A. retroflexus*, *Bunias orientalis*, *Cannabis sativa* var. *spontanea*, *Cirsium arvense*, *Conium maculatum*, *Consolida hispanica*, *Conyza canadensis*, *Cuscuta campestris*, *Digitaria ischaemum*, *Echinochloa crus-galli*, *Galinsoga parviflora*, *G. quadriradiata*, *Iva xanthiifolia*, *Orobanche minor*, *Oxalis corniculata* var. *corniculata*, *O. dillenii*, *Panicum miliaceum* subsp. *agricola*, *Portulaca oleracea*, *Rumex alpinus*, *R. longifolius* subsp. *sourekii*, *Senecio inaequidens*, *Setaria faberi*, *S. verticillata*

An important feature of the Black List is that it adopts a stratified approach to management, which reflects the local and regional context of a particular invasion and allows the formulation of the optimal strategy for each species. An example is the management of *Robinia pseudoacacia* (black locust) in the Czech Republic, the planting of which can be allowed in areas where the stands are not an imminent threat to the landscape, but should be prohibited, and extant stands eradicated, from sites with a nature conservation value, such as dry grasslands (Vítková et al. 2017). The stratified approach thus discriminates where and when the management of alien species is needed and efficient, and where the eradication is neither effective nor necessary, e.g. in urban and suburban areas (Pergl et al. 2016d). The Black, Grey and Watch Lists have served as a basis for implementing the methodology of monitoring alien species by the Ministry of Environment of the Czech Republic (Pergl et al. 2016a).

8.10 Accounts of Invasive Taxa in the Czech Republic

In the following section, factsheets of invasive neophytes in the Czech Republic are presented, as classified in Pyšek et al. (2012b). The distribution maps use the grid cells of the Central European mapping system (KFME, Kartierung der Flora Mitteleuropas; Niklfeld 1998; Schönfelder 1999) and are based on data from the Pladias database (www.pladias.cz). Only occurrences outside cultivation are included in the maps. Maps are not shown for taxa for which insufficient information on distribution is available, which is mostly due to the low reliability of records for taxonomically difficult taxa that are easily mistaken in the field. Note that the distribution maps are generally somewhat underestimated since they are not the result of systematic mapping of invasive species, but for the taxa presented they reflect the patterns in their geographic distribution reasonably well. A complete list of the references from which the information for these accounts was extracted can be found in Pyšek et al. (2012a). For all species these accounts are also partly based on the Flora of the Czech Republic (Hejný et al. 1988, 1990, 1992; Slavík et al. 1995, 1997, 2000, 2004; Štěpánková et al. 2010).

Acer negundo (*Sapindaceae*, Fig. 8.11a) is a dioecious tree reaching a height of up to 20 m; in the Czech Republic it is not as tall being only up to 10 m in height. It is native to eastern and central parts of North America. Its moderate resistance to flooding combined with tolerance to water deficits enable it to occur in a wide range of habitats ranging from wetlands and floodplain forests to relatively dry forests and grasslands. The introduced range covers Europe, parts of Asia and Australia. The species was imported into Europe as an ornamental tree in the seventeenth century, with its first record in the UK dating from 1688. It became a popular garden tree due to its fast growth in the first few years. In the second half of the nineteenth century it was planted in parks, along roads, later on in wind-breaks and shelter belts. The

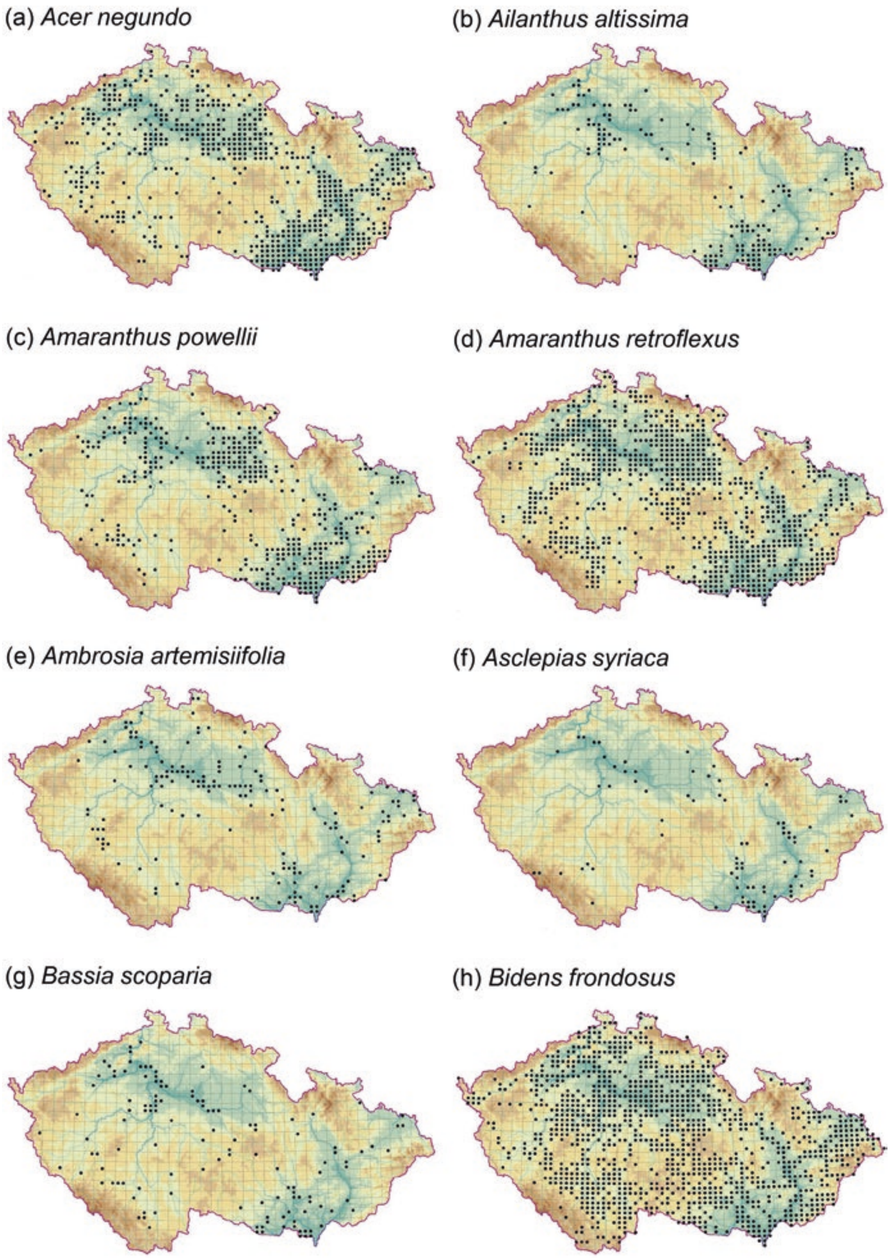

Fig. 8.11 Distribution of selected invasive plant species in the Czech Republic. Species distribution maps in this chapter are based on data stored in the Pladias database (www.pladias.cz, accessed in January 2017), which is used with the permission of the original data providers (Institute of Botany CAS, Masaryk University, Brno, the Nature Conservation Agency of the Czech Republic and Czech Botanical Society), except for *Ambrosia artemisiifolia* which is based on the map published by Skálová et al. (2017). Maps of commonly planted woody species are not shown because their records in the database often relate to planted individuals. All the maps were prepared by O. Hájek

first record of spontaneous occurrence in the wild in the Czech Republic is probably from the bank of the Labe River near Neratovice in 1875. It is commonly grown, with a variety of cultivars on the market. The species is planted in urban areas, industrial zones and wind-breaks, where the planted trees give rise to new populations originating from self-seeding. It spreads by seed and easily resprouts from stumps. It prefers moist open habitats with a sufficient supply of organic nutrients on clayey or sandy soils. Above 350 m a.s.l. it only occasionally reproduces by seed. It also forms spontaneous invasive populations along large rivers, where it invades disturbed floodplain forests (especially along the Morava and lower Dyje rivers), and in areas reclaimed after coal mining. Except in these plantations, it rarely occurs as a strong dominant with a high cover, and often occurs together with other neophytes such as *Fraxinus pennsylvanica*, *Helianthus tuberosus*, *Robinia pseudoacacia*, *Solidago* spp. and *Symphyotrichum* spp. It is recorded from seven habitat types. The invasion is most pronounced in warm areas of the country, namely in southern and central Moravia, where it is common, develops monodominant stands and spreads rapidly. *Acer negundo* is ranked among the 40 most invasive woody species in the world. It has negative impacts due to its pollen causing allergenic reactions in humans and competition with other species at invaded sites.

Ailanthus altissima (*Simaroubaceae*, Figs. 8.11b and 8.12a) is a fast-growing deciduous tree, ~20 m tall, reproducing both vegetatively by root suckers and by seed (up to 325,000 seeds per year are produced by a single tree). It is native to East Asia (China and Korea) and has been introduced worldwide, including Australia and the Pacific Islands. This species was introduced into Europe in 1784 or earlier (some sources suggest an introduction date of ca 1750 into the UK) as an ornamental tree. In the Czech Republic the first record of planting comes from 1813 and in the wild from a forest near Veltrusy, central Bohemia, in 1874. The species is highly tolerant of polluted air and poor soils, and is markedly resistant to varying temperatures, and different levels of humidity, light and moisture and is thus able to grow in stressful habitats. At present it is spreading in large cities and their suburbs and industrial zones where it forms large stands independent of original plantings. Beside urban habitats, it occurs in shrub and grassland vegetation, especially in warmer areas; it is recorded from 11 habitat types (Fig. 8.8). It is spread by wind, water and contaminated soil. As a rather weak competitor it is only able to establish in sparse vegetation, although after establishment it quickly spreads by root suckers. Typical habitats in the Czech Republic are railway corridors, where *Ailanthus* often forms thickets by clonal growth. It often grows along with *Robinia pseudoacacia* stands and together with other neophytes such as *Erigeron annuus*, *Lycium barbarum* and *Parthenocissus inserta*. It is occasionally planted in dry grasslands on steep slopes where it forms clonal scrub or shruby woodland with thermophilous species, vegetation similar to that found in South-eastern Europe. The invasive populations in the Czech Republic occur mainly in southern Moravia. The impact of *A. altissima* on native vegetation is through its formation of dense thickets and allelopathic effects inhibiting growth and germination of native species. The plant sap can induce dermatitis in humans. Another important aspect is that the root system of this tree often damages pavements, walls and buildings.

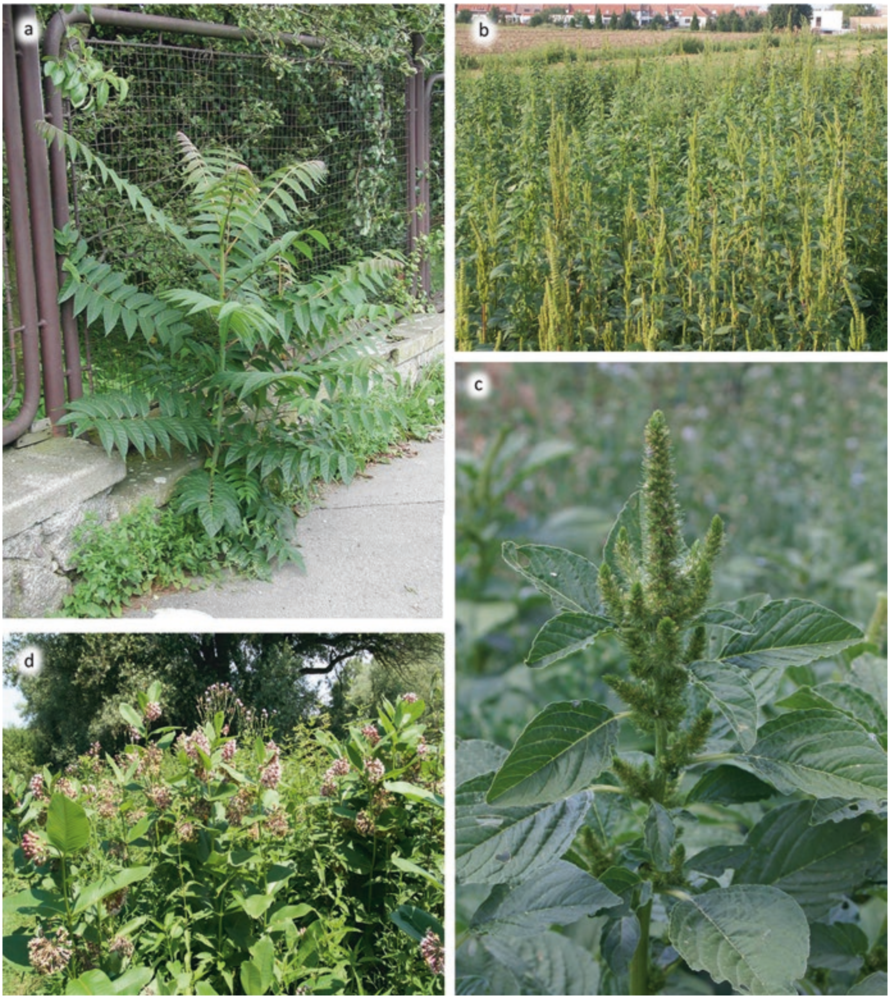

Fig. 8.12 Examples of invasive plant species in the Czech Republic: (**a**) *Ailanthus altissima*, Prague-Troja; (**b**) *Amaranthus powellii*, Lednice, southern Moravia; (**c**) *A. retroflexus*, Lipůvka, southern Moravia; (**d**) *Asclepias syriaca*, Břeclav, southern Moravia. Photo credits: P. Pyšek (a, b), M. Chytrý (c) and Z. Lososová (d)

Amaranthus powellii (*Amaranthaceae*, Figs. 8.11c and 8.12b) is an annual herbaceous plant native to tropical and subtropical South and Central America. Its invaded range includes temperate regions in North America, Europe, Asia, Africa and Australia, where it grows in open ruderal vegetation in urban areas and as a weed in fields and gardens. It was first recorded in the Czech Republic in a potato field near Mnichovo Hradiště, north-eastern Bohemia, in 1853. The earliest herbarium specimens are from 1931. Introduced into the Czech Republic as contaminant in grain, oil seed, ore, cotton and wool, it is common in warm lowland areas and is spreading into colder areas and higher altitudes. Until the 1960s it occurred only in ruderal habitats in villages and along roads. It started to spread rapidly in the 1980s, supported by frequent planting of maize and beet. In the 1990s it became common as a weed on arable land. It is resistant to some herbicides and in the last decade its invasion has been supported by their regular application in agriculture, along railways and in urban areas. Compared to the congeneric *A. retroflexus*, it is less widely distributed, confined to warmer areas, less drought-resistant and more nutrient-demanding, which is reflected in *A. powellii* being currently more invasive in arable land and vegetable gardens than in ruderal habitats. Due to its strong hybridization potential (two hybrids are listed by Pyšek et al. 2012a) it may affect closely related species, although the viability of some of the hybrids is low.

Amaranthus retroflexus (*Amaranthaceae*, Figs. 8.11d and 8.12c) is an annual herbaceous plant native to North America, introduced into regions with temperate and warm climates on all continents. In its native range it is a component of pioneer riverine vegetation. The earliest records in the Czech Republic, where it was introduced as a contaminant in grain, oil seed, cotton and wool, are from 1818 in Prague and 1822 in Uherské Hradiště, south-eastern Moravia. Seed contamination of soils and commodities is linked to its extreme fecundity, with a single plant producing up to 500,000 seeds that remain viable in the soil for up to 5 years. It grows in relatively dry and nutrient-rich urban and agricultural habitats such as rubbish dumps, soil heaps after digging along roads, rivers and railroads, and as a weed on arable land; it is recorded from 10 habitat types. Its invasion has been supported by an increase in maize and beet planting, although currently it benefits mostly from its tolerance to herbicides, high salinity levels, polluted soil and eutrophication of the landscape. Further rapid spread is unlikely since the species is widespread and common and appears to have colonized most available habitats. However, further spread of herbicide-resistant populations in less suitable habitats as well as into higher altitudes is possible. The major impact of this species is a reduction of crop yield. It hybridizes with some species from the *A. hybridus* agg., including cultivated ornamentals or pseudocereals.

Ambrosia artemisiifolia (*Asteraceae*, Fig. 8.11e) is an annual herbaceous plant that is native to North America, including the central and eastern USA, where it grows as a pioneer species in open semi-arid habitats. In its invaded range, which includes all continents (except Antarctica) as well as some islands (New Zealand, Hawaii, Madagascar and Mauritius), it is known from a wide range of open and nutrient-rich, disturbed ruderal habitats and arable land. This species was introduced into Europe in the second half of the nineteenth century as a contaminant of agricultural commodities, bird seed and with agricultural machinery. Several independent introductions are documented. Early introductions into Europe mostly resulted in short-lived casual occurrences. Established populations only developed in the first decades of the twentieth century, and commonly after World War II. Within the last few decades *A. artemisiifolia* has significantly increased its range and abundance in many European countries, and spread into a number of habitats. The invasion is supported by the production of a high amount of seed (1200–2500 seeds per plant) that forms a long-term persistent soil-seed bank, with seed remaining viable for up to 40 years. The first record in the Czech Republic is from 1883, in a clover field near Třeboň, southern Bohemia, and from a field near Plzeň-Doudlevce, western Bohemia. The seed was probably introduced with clover seed from North America. The next wave of introductions occurred in the second half of the twentieth century from different sources: with grain from Canada and with soybeans from North America, and with Ukrainian grain and Soviet ore. This species is confined to the warm parts of the Czech Republic and prefers open dry habitats on sandy or gravelly substrata with a low cover of vegetation. After rapid spread in the 1980s–1990s the invasion has decelerated. *Ambrosia artemisiifolia* is limited by requiring well-aerated soils and being a weak competitor. It rarely spreads in urban spaces, in sand pits, on coal mining heaps or in semi-natural grasslands. In southern Moravia, however, it has started recently to occur as a weed in maize fields. It grows mostly in ruderal habitats, in particular along railways where it is supported by the use of herbicides, to which some populations are resistant. This resistance is due in part to the fact that herbicides are applied at the end of spring when *Ambrosia* populations are only starting to germinate as their phenological optimum is in autumn. It is scattered and locally abundant along railways but does not yet form monodominant stands. The pollen of *A. artemisiifolia* is the most allergenic of all plant species occurring in Europe. Ragweed pollen peaks in August and September in the Czech Republic, especially in southern Moravia. It is spread by easterly winds, most probably from the Hungarian lowlands. Due to its so far rare occurrence in the Czech Republic it does not present a serious problem for the allergic population in this country, but its importance as an allergen is likely to increase in the future. The economic costs of ragweed invasion in Germany are estimated at 32 million EUR annually, nearly entirely incurred within the health sector. Furthermore, in other countries *A. artemisiifolia* significantly reduces crop yields, especially in spring-sown crops like sunflower, soybean, maize and vegetables. However, in most of Europe, the infestation of agricultural fields is a relatively recent phenomenon, thus impacts on crop yield are still minor.

Asclepias syriaca (*Apocynaceae*, Figs. 8.11f and 8.12d) is a rhizomatous perennial native to eastern North America, with a distribution ranging from Canada to North Carolina. In its native range this species grows in prairies, along roads and railways and at disturbed sites. It has been introduced into many parts of the world, and the introduction into Europe dates back to 1629. The first record of its planting in the Bohemian lands is from the Lány chateau park, central Bohemia, in 1785, and it escaped from cultivation already before 1821. It was planted for multiple purposes in the past, including medicinal, textile, oil and honey production. It is currently cultivated mostly as an ornamental. In the Czech Republic it is abundant and invasive only in warm lowlands, especially in southern Moravia, where it spreads over long distances by seed. However, the spread by rhizomes is more important, such as with contaminated soil during railway construction. In Bohemia, it forms rather large but isolated stands, and long-distance dispersal is limited. Populations occur in open habitats along roads, railways, in vineyards and shruby margins, abandoned places in settlements and on dry banks of streams. The tall stands of *Asclepias syriaca* persist for decades and are relatively species-rich. The species is toxic to humans and herbivores. Some impact of *Asclepias* on species diversity is reported from Southern Europe where it colonizes nutrient-poor soils and sand dunes.

Bassia scoparia (*Amaranthaceae*, Fig. 8.11g) is an annual plant up to 2 m tall, native to a large area from South-eastern Europe to East Asia, where it grows in dry open steppe habitats and as a weed in arable fields. It has been introduced as a contaminant of crop seed and an ornamental plant into rather warm parts of Europe, South Africa, both Americas and Australia. Within this extremely variable species, two poorly defined subspecies are traditionally distinguished in the Czech Republic, **subsp. *scoparia*** and **subsp**. ***densiflora***. The first report of cultivation of the subspecies *scoparia* in the Czech Republic dates back to 1811 and that of its escape to 1819. Plants referable to subspecies *densiflora* were first collected in Moravia in Popice in 1901 and in Bohemia in Praha-Zlíchov in 1930; later they were introduced with commodities from the former Soviet Union. The rapid spread in warm areas only started in the early 1990s. This species prefers sandy and gravelly soil poor in nutrients, and occurs in open dry habitats; it grows well on salty and polluted soils. *Bassia scoparia* colonizes ruderal habitats, namely along railways, but it also grows along roads, in sand pits and on sand heaps, forming dense closed stands, which persist for several years. It is wind-pollinated, thus maintaining a genetic link with populations in cultivation. In southern Moravia it is reported to occur as a weed in maize fields. Populations are herbicide-resistant and have their development peak in the autumn, therefore they are not harmed by the spring application of herbicides. Both subspecies occur together at the same localities, but the subspecies *scoparia* is more common.

Beta vulgaris **Altissima Group** (*Amaranthaceae*) includes, in addition to sugar beet, also annual weedy types that were first recorded in the 1980s in the Czech Republic. These plants were introduced with beet seed from Southern and Southwestern Europe, mainly Italy and France, where they originated from the pollination of cultivated sugar beet (*B. vulgaris* Altissima Group) by the pollen of the wild *B. vulgaris* subsp. *maritima* or of weedy annual plants derived from some cultivars of the Altissima Group. The first records of this "weedy beet" come from the 1970s in the UK. It started to spread very fast only rather recently. A survey from 2006 revealed that weedy beet occurred on 70% of the farms in the Czech Republic growing sugar beet and on 4% of those its density exceeded 1000 plants/ha. Plants of weedy *B. vulgaris* occur in a variety of growth forms, from large to small roots and differing in root branching, total plant weight or in the number seeds they produce. They germinate early in the season before cultivated sugar beet. The seeds form a long-term persistent seed bank, which makes eradication difficult. Occurrence of weed beet in fields is a serious economic problem as it competes with sugar beet and makes it more difficult to harvest and process sugar beet. The chemical control of weedy beet is prevented by the impossibility of using herbicides in sugar beet fields. Alternative (mechanical) methods can partly reduce the seed set, but high regeneration capacity makes it necessary to repeat the cutting of flowering stems several times in a season. Hybridization with native species in the Czech Republic has not been reported, but transfer of genes from GMO sugar beet to weedy species was observed.

Fig. 8.13 Examples of invasive plant species in the Czech Republic: (**a**) *Bidens frondosus*, Cep sandpit near Suchdol nad Lužnicí, southern Bohemia; (**b**) *Bunias orientalis*, Pouzdřany, southern Moravia; (**c**) *Conyza canadensis*, a railway bank near Všetaty, central Bohemia; (**d**) *Echinocystis lobata*, Sázava River bank in Sázava, central Bohemia; (**e**) *Echinops sphaerocephalus*, Kobylí hlava hill near Hluk, southern Moravia. Photo credits: M. Chytrý

Bidens frondosus (*Asteraceae*, Figs. 8.11h and 8.13a) is an annual plant native to large areas in North America, from southern Canada to the southern USA, where it grows in riparian habitats, on lake shores, as well as along roads and railways. The invaded range covers Europe, Asia and New Zealand. It was introduced into Europe, possibly as a contaminant in wool, in the late eighteenth century, with the first record from 1777 in Poland. In the Czech Republic it was first reported in 1907 and became naturalized in the first half of the twentieth century. At present it occurs in most parts of this country and colonizes a wide range of moist and nutrient rich habitats. It is recorded in 21 habitat types, and is a dominant species in vegetation consisting of nitrophilous annual hygrophilous plants (Fig. 8.8). Its competitiveness with native *Bidens* species is enhanced by its markedly higher drought resistance. It occurs on riverbanks, shores of water bodies, in road ditches and moist waste places. This species is common in urban areas, industrial zones and mining areas. It forms dense but small populations, usually up to several m^2. *Bidens frondosus* has little impact on the diversity of invaded communities; early germination and tall stature enable it to suppress native species with which, nevertheless, it coexists following invasion. It appears to have colonized the majority of suitable habitats, and further spread thus depends on the frequency of available sites.

Bunias orientalis (*Brassicaceae*, Figs. 8.13b and 8.14a) is a biennial or perennial herbaceous plant up to 1.7 m tall, reproducing by seed and root fragments. Its native range is in Siberia and Eastern and South-eastern Europe, although according to some authors it was originally restricted to Armenia. It grows at the edges of forests and along riverbanks. It has been introduced into North America and most of Europe, where it has been known since the seventeenth century. The first record in the Czech Republic is from 1856. It was introduced as a contaminant in maize seed and with horse fodder imported from Russia. Until the 1920s there were only a few occurrences reported in Europe, but it became more common in the Czech Republic after World War II. Recent spread is often due to unintentional transport with soil. It has spread very fast in the last two decades, mainly in grassland along roads and railways, in abandoned areas, on reclaimed mining areas in the lowlands and moderately warm mid-altitudinal regions. Only recently it started to extend its altitudinal range and is also spread by floods. It occurs in a range of ruderal, but also semi-natural habitats, assigned to nine habitat types. Optimum conditions include perennial thermophilous ruderal vegetation on deep and dry clayey soils rich in mineral nutrients. This species mostly invades secondary human-made habitats, where the impact on native diversity is relatively low; however, it may become a troublesome weed in some agroecosystems. It can reduce species diversity in grasslands, in which it has developed large and dense populations in the last 20 years.

Cannabis sativa* var. *spontanea (*Cannabaceae*, Fig. 8.14b), an annual herbaceous plant up to 1.3 m tall, native to dry steppe areas in Central Asia, where it also grows at disturbed sites along roads and railways, and in cities and villages, i.e. in habitats similar to where it grows in its invaded range in Europe and North America. This taxon has been introduced into Europe for use as bird seed and as an agricultural crop used for fibre and was also unintentionally introduced as a crop contaminant. In the Czech Republic it was first recorded growing in the wild in 1868, at

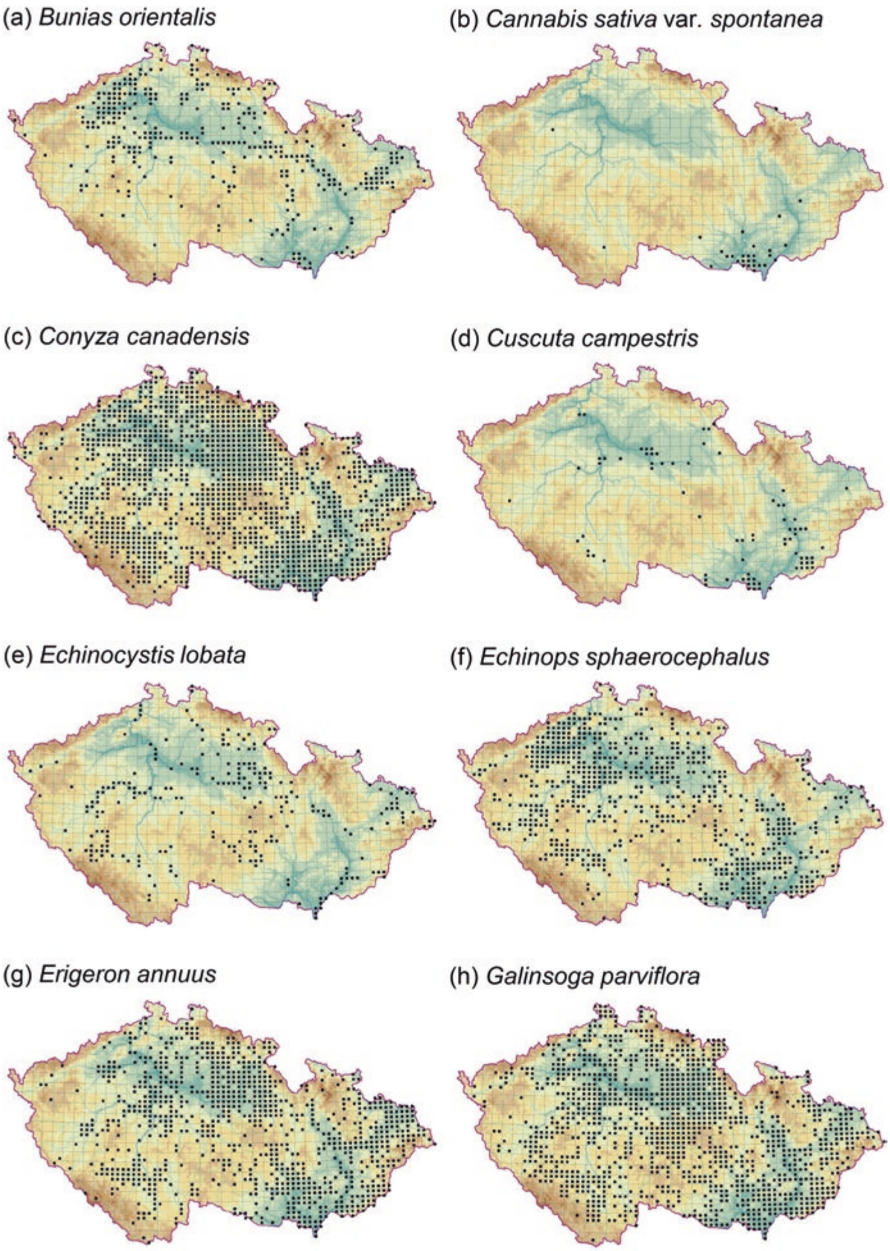

Fig. 8.14 Distribution of selected invasive plant species in the Czech Republic

Hustopeče in southern Moravia. Its distribution in the Czech Republic is confined to the warm and dry areas in southern Moravia, where it spreads in agricultural landscapes and in the surroundings of villages on deep, dry soils rich in nitrogen and mineral nutrients. It invades annual vegetation in ruderal habitats and on arable land, and produces dense stands on rubbish dumps, waste places, along roads and paths, and at the margins of vineyards, maize and other fields. Hybridization between *C. sativa* var. *spontanea* and cultivated forms is reported.

Conyza canadensis (*Asteraceae*, Figs. 8.13c and 8.14c) is an annual species native to North America where it grows at disturbed open patches in meadows or fields. A single plant produces up to 100,000 cypselae. The species has been unintentionally introduced into all other continents except Antarctica. It is common, especially in regions with temperate and subtropical climates. Typical habitats in the invaded range are dumps, ruderal and urban sites, road and railway verges or disturbed grasslands. The first record in the Czech Republic is from 1750. It is reported as naturalized and common in early floras from the late eighteenth and the early nineteenth century, while numerous records from the beginning of the eighteenth century exist for neighbouring Germany. It is widely distributed in the Czech Republic, and occurs in 33 habitat types; it is a dominant in annual ruderal vegetation and thrives in many other habitats (Fig. 8.8). Although it was introduced a long time ago, it is still spreading into higher altitudes and semi-natural vegetation, forest clearings and other habitats. Its invasion is supported by its resistance to triazine-based herbicides, which is reported to have developed in the 1970s, and which enables it to spread in urban areas or industrial zones. It grows as a weed in vineyards, hop fields, orchards and vegetable fields. In Europe, it is a significant weed in warmer areas and in fields where it may reach high densities and therefore negatively affect crop yield.

Cuscuta campestris (*Cuscutaceae*, Fig. 8.14d) is an annual parasitic plant native to western North America where it grows in open grasslands and fields. It has been introduced into most of the world, often as a contaminant of seed. The first record in Europe is probably from Prague in 1883. The distribution of this species in the Czech Republic is scattered. It is common only in warm lowlands such as those of the Labe River and in southern Moravia, where it usually occurs in wasteland along roads and also as a weed in clover and alfalfa fields and gardens, or in southern Bohemia, where it grows on the bottoms of drained fishponds. It only occurs in four habitat types and as a dominant in annual and perennial nitrophilous ruderal vegetation; its recent spread is attributed to its high tolerance of salinity, a feature that enables it to spread along roads. Where it is abundant it has a major impact on agriculture in that it reduces yield. It was considered as rare in the 1960s; at present it is starting to extend its altitudinal range.

Echinocystis lobata (*Cucurbitaceae*, Figs. 8.13d and 8.14e) is an annual vine native to North America where it grows at open sunny sites in floodplains and at the fringes of forests. This species was introduced into temperate and continental Europe in the early twentieth century (1906, Slovakia) and into the Bohemian lands in 1911. Its introduction into Europe is linked with botanical gardens, from where it spread across the continent. The frequency of planting has increased in the last 20 years, which accelerated the spread of this species. Its invasion in the Czech Republic was supported by seed transported by water, especially during floods, and it is still actively spreading via water in Eastern Europe. This mode of dispersal is reflected by its distribution in the Czech Republic, where it is confined to areas along large rivers. It invades semi-natural habitats, being assigned to seven habitat types, with its ecological optimum in lowland nitrophilous herbaceous fringes, willow galleries on riverbanks and perennial nitrophilous vegetation at mesic to wet sites. This species is demanding of light, nutrients and moisture, which limits its invasion to the vicinity of rivers. The impact of this species is through its ability to cover large areas and overgrow native vegetation. The whole plant contains substances that are toxic to humans and animals.

Echinops sphaerocephalus (*Asteraceae*, Figs. 8.13e and 8.14f) is a tall herbaceous perennial that is native to Europe, with its native distribution ranging from Southern Europe to Southern Siberia. It occurs as an introduced species in the rest of Europe and North America. In both distribution ranges it grows at disturbed sites along roads and rivers, in urban areas and dry grasslands, preferring nutrient-rich soils and sunny places. In the Czech Republic it is grown as a garden ornamental and is still sown in the wild by bee keepers, e.g. on railway banks. The first record from the Czech Republic is from 1871. This species is relatively widely distributed, being more abundant in warm areas and occurs in nine habitat types and thrives best in perennial thermophilous ruderal vegetation. It has recently spread in dry, disturbed ruderal grasslands. In the past its occurrence was restricted to the surroundings of villages and stone quarries, while currently it is common in suburban areas and spreads in the landscape along roads and is extending its altitudinal range. It also invades mesic habitats such as road ditches. No clear impacts on biodiversity or human health are known.

Erigeron annuus (*Asteraceae*, Figs. 8.14g and 8.15a) is native to North America (northern and eastern USA and south-eastern Canada) where it grows in dry forests, forest clearings, open rocky sites, grasslands, along roads and railways, and as a weed in gardens and fields. Its introduced range covers Europe, and parts of Asia and New Zealand. Introduced into Europe as an ornamental in the early eighteenth century, its later spread across the continent was mostly due to unintentional introductions as a seed and soil contaminant. The earliest record of this species in the Czech Republic is from 1884. There are two subspecies differing in their distribution in the Czech Republic: **subsp. *septentrionalis*** is more common, while **subsp. *annuus*** is scattered. However, its precise distribution and invasion history is difficult to outline as both subspecies are easy to confuse, making literature reports less reliable. It grows in 12 habitat types, with an ecological optimum in the herbaceous fringes of lowland rivers, forest clearings and perennial thermophilous ruderal vegetation (Fig. 8.8), and is occasionally planted in gardens. It prefers light and dry habitats, with its recent spread supported by large-scale disturbances, eutrophication and extension of suburban areas. In the second half of the twentieth century it was mostly an urban weed limited to gardens and waste places but currently it is present in open landscapes, where it occurs not only at disturbed sites but also in

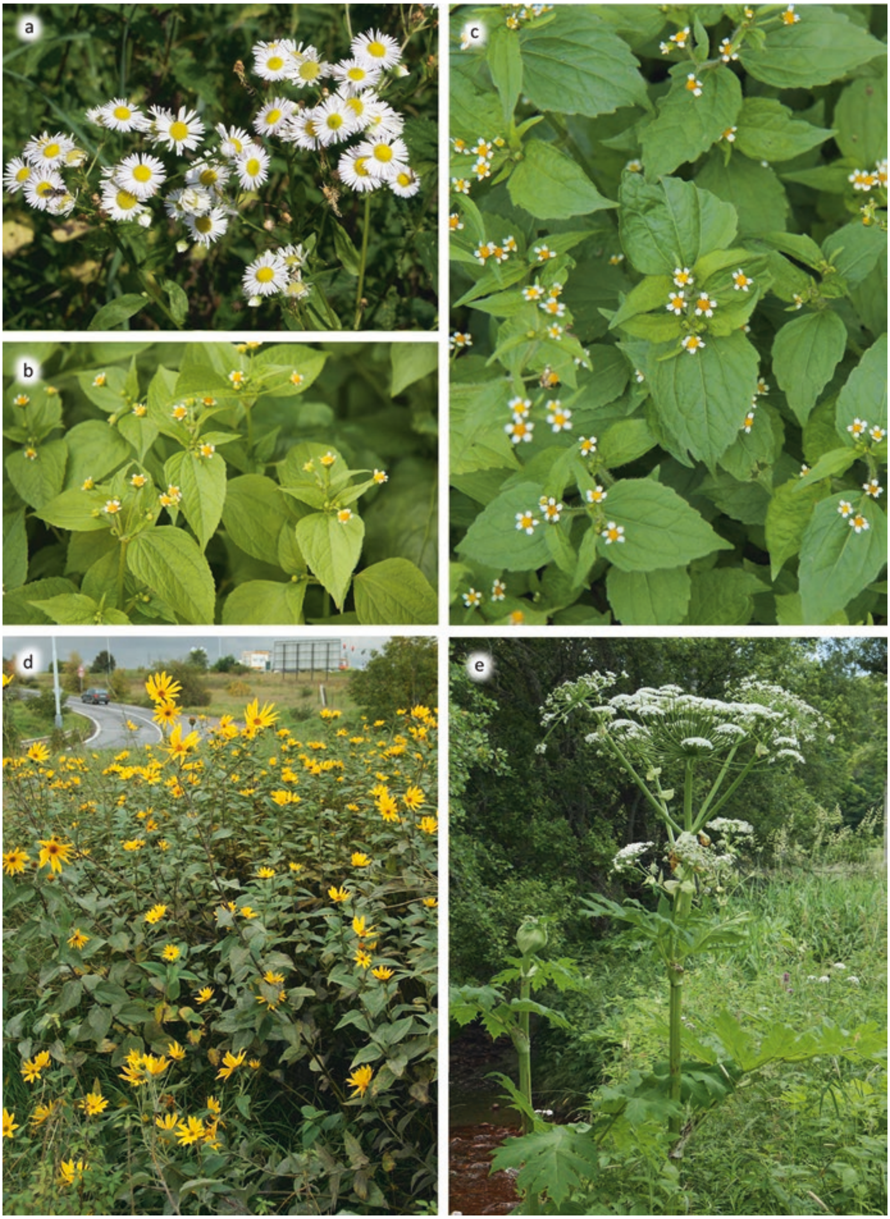

Fig. 8.15 Examples of invasive plant species in the Czech Republic: (**a**) *Erigeron annuus*, Klecany, central Bohemia; (**b**) *Galinsoga parviflora*, Brno; (**c**) *G. quadriradiata*, Ketkovice, southern Moravia; (**d**) *Helianthus tuberosus*, Prague; (**e**) *Heracleum mantegazzianum*, Františkovy Lázně, western Bohemia. Photo credits: P. Pyšek (a) and M. Chytrý (b–e)

semi-natural grassland. Its invasion is supported by disturbance of the soil surface by occasional mowing. The impact of this species is as a weed of arable land.

Fraxinus pennsylvanica (*Oleaceae*) is a deciduous tree up to 25 m tall, native to eastern North America, where it is a light-demanding and early successional species in a wide range of environmental conditions, although it mostly occurs on moist and nutrient-rich soils along rivers. The first record of its planting in Europe is from 1783, and in the Czech Republic from 1835. This species is planted in towns and occasionally also in forests. It spreads into natural and semi-natural alluvial forests, where it thrives best. It also occurs in scrub and pioneer woodlands in forest clearings and spreads in suburbs and coal-mining areas, where it is planted for habitat reclamation. As a fast-growing and early-reproducing woody plant, it establishes in ruderal vegetation in waste places early in succession and later invades grassland, scrub and forest margins. Its distribution is scattered, confined mainly to large river floodplains. At present it is spreading fast but its invasion is in the initial phase.

Galinsoga parviflora (*Asteraceae*, Figs. 8.14h and 8.15b) is an annual species native to the South American Andes, with secondary distribution in temperate and subtropical regions. It is recorded growing from the lowlands up to 3600 m a.s.l. As for its congener *G. quadriradiata*, its natural habitat in its native range are floodplains. In the invaded range it occurs as a weed in arable fields and gardens as well as in disturbed urban habitats. A single plant can produce up to 6000 highly germinable cypselae. The earliest record of *G. parviflora* in Europe is from 1785 at a botanical garden in Paris, France. In the Czech Republic it was first reported in 1880 and records started to accumulate rapidly; it spread very quickly from the 1920s to the 1940s, when this species became common. Following the initial introduction into botanical gardens and subsequent escape, the main pathway of further spread was as a contaminant of soil and crop seed. This species occurs in a wide range of habitats in the Czech Republic (13; Fig. 8.8), however, in most of them only occasionally. It requires well-aerated or loose, moist and nitrogen-rich soil, but cannot tolerate a high cover of other species and is sensitive to frost; these factors limit its colonization of sites at higher altitudes. It is abundant in gardens, on rubbish and compost heaps, along walls in cities and in root-crops. In the Czech Republic it is common and has colonized the majority of sites with suitable habitats. Recently, for example, it has started to appear in road verges. This species spreads both by seed and clonally by adventitious roots at the stem base; if weeded manually, rooting bases of stems remain in the soil and ripening of cypselae continues for a few days after removal. These features make it a serious weed in crops. It is a host plant of some agriculturally important viruses, insects and nematodes.

Galinsoga quadriradiata (*Asteraceae*, Figs. 8.15c and 8.16a) is an annual plant native to South and Central America, where it occurs in periodically flooded sites along rivers. As for its previously described congener, it has been introduced into other continents where it occurs in similar human-made habitats. It was introduced into Europe in the nineteenth century. The earliest record of planting in the Czech Republic is from 1823 and that of escaped plants from Prague in 1890. Both species are very similar in terms of pathways of introduction and dispersal, ecological requirements and habitat affinities. The rapid spread of this species occurred later than that of its congener, during the second half of the twentieth century. Currently

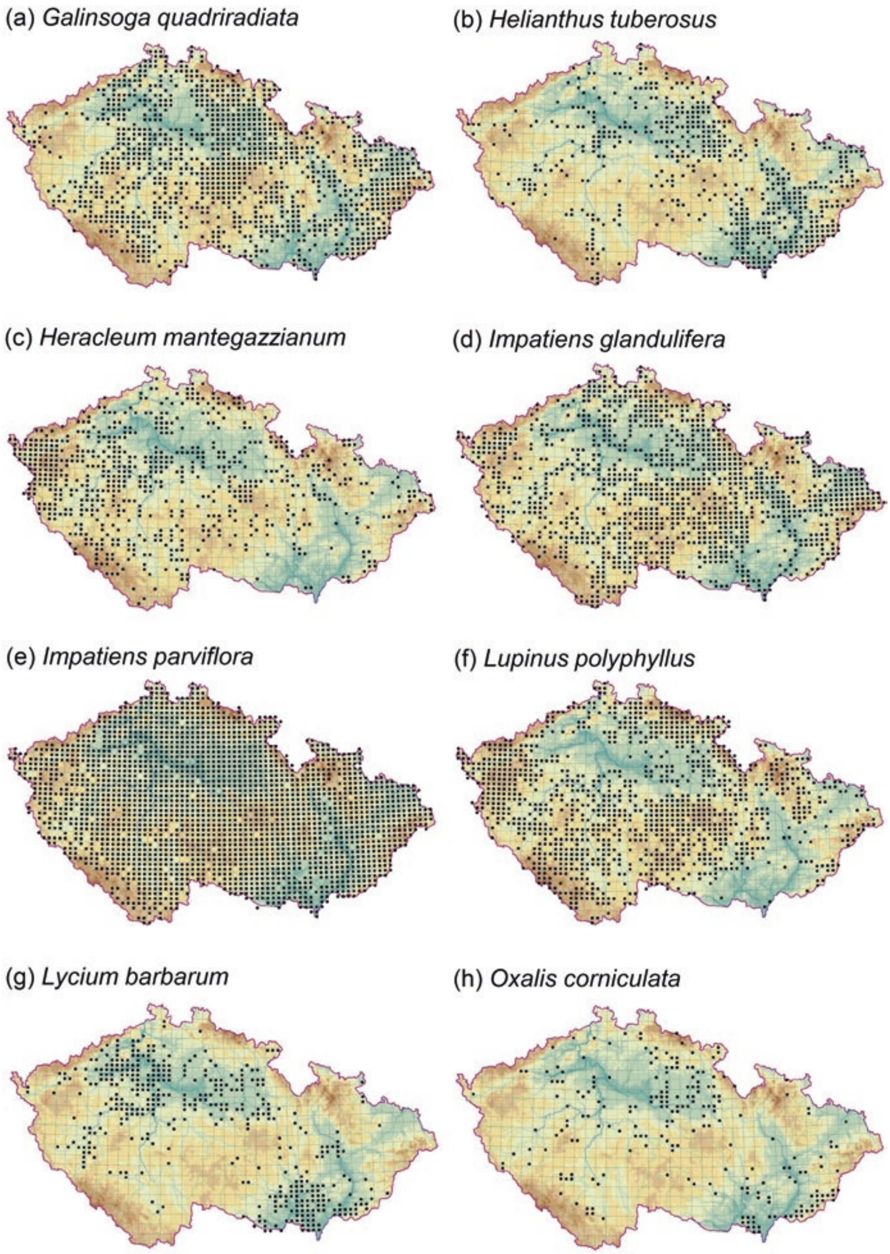

Fig. 8.16 Distribution of selected invasive plant species in the Czech Republic

it is less common than *G. parviflora* but also occurs in most of this country. It is tolerant of high concentrations of salt and heavy-metals. Its impact is that of a serious weed in agricultural crops and as a host plant of some agriculturally important viruses, insects and nematodes.

Helianthus tuberosus (*Asteraceae*, Figs. 8.15d and 8.16b) is a perennial herbaceous plant up to 3 m tall, reproducing by seeds and tubers. This species is a native to the central and eastern parts of the USA and south-eastern Canada, where it grows in wet meadows and abandoned fields. In its invaded range (parts of North and South America, northern Africa, Australasia, temperate Asia and Europe) it occupies open ruderal sites along roads and rivers as well as field edges and urban habitats with nutrient-rich soils. This species was introduced into Europe probably in the seventeenth century. The first record in the wild in the Czech Republic dates back to 1885. *Helianthus tuberosus* is planted for ornamental purposes and as food for wild animals, namely wild boar. Recently, the tubers started to be more popular as dietetic food. This species prefers clayey, humid, nutrient-rich soils and spreads locally near plantings, such as in villages and forest openings. It is, however, most invasive in floodplains where tubers are spread by floods; on fluvial sediments tubers can sprout from a depth of up to 1 m. It is invasive particularly along large rivers in Moravia and in the surroundings of lowland settlements. *Helianthus tuberosus* is assigned to seven habitat types and is dominant in nitrophilous herbaceous fringes of lowland rivers, but also thrives in perennial ruderal vegetation at warm and mesic sites. Primary (F1) hybrids with other species of the genus are not recorded. The probability of their occurrence is low due to the late flowering of *H. tuberosus*, which prevents the seed from ripening under local conditions. Impact on the species diversity of invaded communities is reported from the Czech Republic: stands reaching 50–100% cover reduce species richness by ~30%.

Heracleum mantegazzianum (*Apiaceae*, Figs. 8.15e and 8.16c) is usually monocarpic and short-lived perennial species which can live up to 13 years, reproducing exclusively by seed, forming a short-term persistent seed bank lasting for at least 5 years and persisting in some localities for decades. A single plant produces on average 20,000 seeds of which up to 90% germinate. It is native to the Western Greater Caucasus, where it grows in tall-forb vegetation below the tree line, on forest clearings and along forest margins. In 1817 it was introduced into Europe as a garden ornamental and multiple introductions followed. Now it is considered as invasive or naturalized in many European countries and North America. The invasion of *H. mantegazzianum* in the Czech Republic is recorded in detail. The first record comes from 1862, when it was introduced into the park of the Kynžvart chateau in western Bohemia. Fifteen years later it was found to have escaped and naturalized in the close vicinity of this garden. The duration of the lag phase, or the time between its introduction and the start of its exponential spread, is estimated as ~60 to 70 years, with the rapid invasion starting in the 1940s. Rapid spread and increase in abundance of *H. mantegazzianum* was promoted by a radical change in land use and human-mediated disturbances after World War II, especially in the west of the country, where the species was originally introduced. *Heracleum mantegazzianum* invades nutrient-rich sites in semi-natural grasslands, forest edges and anthropogenic habitats. However, it is also able to establish in nutrient-poor habitats, such as fen meadows or forest clearings on acidic soils. Based on local conditions, it can form large populations of thousands of individuals; however, more often it is found in smaller populations of few individuals along linear landscape features such as roads and streams. In the Czech Republic, it occurs mainly in western Bohemia, from where it spread eastwards. In other parts of the country it forms mostly small stands. It rarely occurs in dry and warm lowlands. *Heracleum mantegazzianum* is reported to reduce species diversity of invaded communities; stands with 70–100% cover reduce species richness by 50–60%. This species is harmful to humans due to its phytotoxic sap causing blisters on the skin. It is difficult to eradicate due to the existence of a seed bank and high regeneration ability.

Impatiens glandulifera (*Balsaminaceae*, Figs. 8.16d and 8.17a) is an annual species native to the Himalayas, reaching up to 2.5 m in height and one of the most invasive species in Europe. It was introduced as a garden ornamental into Europe (UK) in 1839 and first recorded as escaped in 1855. Currently it is recorded in 35 European countries. It has also been introduced into North America. In both distribution ranges it grows on disturbed riverbanks, in roadside ditches and at forest edges. In the Czech Republic, the first record of its planting as a garden ornamental comes from 1846 and that of an occurrence outside cultivation from 1896. However, rapid invasion only started in the mid-twentieth century and currently the species is common in this country but only rarely cultivated. *Impatiens glandulifera* is a dominant species in the nitrophilous herbaceous fringes of lowland rivers. It also finds optimum conditions in willow galleries on loamy and sandy river banks and in riverine reed vegetation, and it occurs in 16 habitat types. It also invades fresh soil heaps, forest clearings, forest edges on slopes and forest road verges. Until recently its invasion in the Czech Republic was restricted to the floodplains and surroundings of villages with nutrient-rich humus and permanently moist soils. However, in the last few decades this species has started to widen its habitat niche by spreading outside floodplains, such as in forest clearings and abandoned meadows. At present, it is beginning to colonize drier sites, which in some cases are less rich in nutrients and shaded by trees. This species produces a higher biomass than its congeners and is plastic in terms of its response to nutrient availability and shading, and also exhibits some genetically based population differentiation. Its competitive ability in the Czech Republic may be reduced by late frosts, to which seedlings are sensitive. It regenerates well after disturbance by means of adventitious roots growing from stem nodes. Due to its rapid spread and extensive populations in riparian habitats, it is considered a conservation problem. However, despite forming populations with a cover of up to 90%, it does not markedly reduce the numbers of species co-occurring in invaded stands, although invasion does alter species composition. *Impatiens glandulifera* is also known to reduce the availability of pollinators for co-occurring native species.

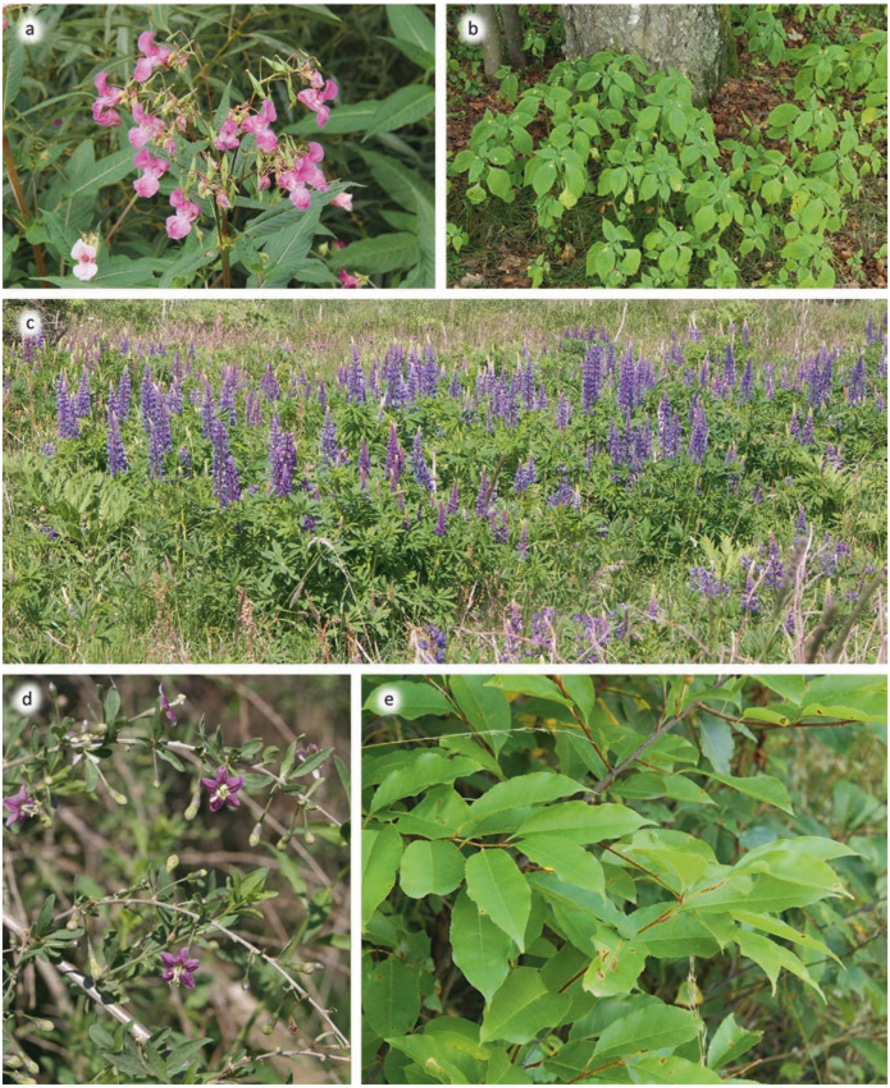

Fig. 8.17 Examples of invasive plant species in the Czech Republic: (**a**) *Impatiens glandulifera*, Radošov, western Bohemia; (**b**) *Impatiens parviflora*, Aš, western Bohemia; (**c**) *Lupinus polyphyllus*, Slavkovský les Mts, western Bohemia; (**d**) *Lycium barbarum*, Výhon hill near Blučina, southern Moravia; (**e**) *Prunus serotina*, Hory, western Bohemia. Photo credits: M. Chytrý (a, b, d, e) and P. Pyšek (c)

Impatiens parviflora (*Balsaminaceae*, Figs. 8.16e and 8.17b) is an annual herbaceous plant assumed to be native to the mountains in Central Asia (including the southern part of western Siberia, western Mongolia, the adjacent Turanian region and the western Himalayas). Its habitats in its native range include shaded streamsides and stony mountain slopes. In its introduced range, it is widely distributed in Europe, Africa, North America and Asia; the first record in Europe is from 1831 from a botanical garden in Geneva. In the Czech Republic it was first recorded in 1844 in a botanical garden in Prague, and in the wild in about 1870. It became widely distributed in the Czech Republic after World War II, with the period of its most rapid spread occurring in the 1970s and 1980s, when it dominated the understory in both natural and cultivated forests. Subsequently, its invasion was slowed by an introduced monophagous aphid, *Impatientium asiaticum*. At present, *I. parviflora* is common all over the country except in treeless landscapes or nutrient-poor coniferous forests. It is, however, less dominant than three decades ago and appears to have already colonized the majority of suitable habitats. *Impatiens parviflora* is strongly confined to sites shaded by trees, preferring loose, moist humus- and nutrient-rich soils, and invading both disturbed ruderal habitats in and around settlements and more natural forest habitats. It is recorded from 45 habitat types and is a dominant in a number of them (perennial nitrophilous herbaceous vegetation at mesic sites, alluvial forests, oak-hornbeam forests and ravine forests) and a frequent dominant in *Robinia pseudoacacia* plantations (Fig. 8.8). In the Czech Republic *I. parviflora* is reported to be less plastic in terms of its response to nutrients and shading than its congener *I. glandulifera*, but exhibits stronger genetically based population differentiation. It is also highly sensitive to frosts. Its impact on native biodiversity in the invaded range is probably weak because of its poor competitive ability.

Lupinus polyphyllus (*Fabaceae*, Figs. 8.16f and 8.17c) is a rhizomatous perennial up to 1.6 m tall, native to western North America, where it grows in wet mountain grasslands and along streams. It has been introduced and became naturalized in the eastern parts of North America and in Europe, including its northern part. In Europe it was first recorded in the UK in 1826 and in the Czech Republic in 1895. It is planted as a garden ornamental, but the invasion was more promoted by it being sown in the wild. From the late nineteenth century it was sown, as a nitrogen-fixing plant, in forests for enrichment and amelioration of acidic soils and as a pasture crop for animals. It is still used for stabilizing the soil of banks along roads and railways. It differs from most other invasive herbaceous plants in the Czech Republic in being confined to non-calcareous, slightly moist and nutrient-poor soils in cold hilly landscapes and foothills. A genetic link with ornamentals cultivated as *L.* ×*hybridus* is probable though not proven. This species is common in the Czech Republic, especially in the west. It is recorded in 14 habitat types and can be dominant in perennial thermophilous ruderal vegetation (Fig. 8.8). It invades unmanaged grasslands along road sides and railways, at ruderal and disturbed sites, in meadows and the margins of forests, or on sandy soils. However, at present *L. polyphyllus* exerts its main impact by dominating vegetation in game preserves, former military areas and slag heaps. Despite forming stands with a cover of 60–95% in invaded vegetation, its effect on native species richness is rather moderate, with ~20% reduction being reported.

Lycium barbarum (*Solanaceae*, Figs. 8.16g and 8.17d) is a shrub up to 3 m tall. Some authors consider it to be a native to South-eastern Europe and Asian parts of the

Mediterranean area, but it is more likely a native to China. It has been introduced into other parts of Europe and Asia, northern Africa and North America. The first record of planting in the Czech Republic is from 1785 and that of its occurrence in the wild is from 1870. This species is rather common, being more abundant in warm areas. Its distribution to a large extent still corresponds to plantings along railways in the nineteenth and early twentieth century. In warm and dry regions it was also planted in hedgerows, as an ornamental shrub and for soil stabilization in the past, but currently is only rarely used. It is a deep-rooting, light-demanding species, colonizing deep, relatively dry, nutrient-rich basic soils at exposed or disturbed sites. Most stands originate from planted shrubs that spread by vigorous clonal growth; it does not reproduce by seed in the Czech Republic as it rarely bears fruit and seedlings do not establish. Long-distance dispersal is by root fragments and rooting branches that get dispersed following disturbance, e.g. during the remodelling of railway corridors. Populations last for a long time and are supported by cutting, which reduces competition from co-occurring trees and initiates new growth. It occurs as a dense stout shrub, a climber in stands of trees, or a small shrub on extremely dry substrates. It grows in 11 habitat types (Fig. 8.8). The invasion of the Czech Republic seems to have reached its peak: the populations persist and increase in size, but new stands rarely develop. The ability of *L. barbarum* to rapidly occupy new sites and exert a strong impact on native diversity is documented for other parts of Europe.

Oxalis corniculata (*Oxalidaceae*, Fig. 8.16h) is an annual, biennial or short-lived creeping perennial herbaceous plant with a native distribution range in the tropical and subtropical belts of Asia and Africa, including the Mediterranean region. It has been introduced into the temperate zones of the Northern Hemisphere (Europe, North America and Asia) where it became naturalized in open disturbed sites along roads, in the edges of fields and urban gardens. It occurs as a serious weed in gardens, fields, lawns and glasshouses. In the Czech Republic it was first reported in 1852 and is most likely still being introduced with contaminated soil. It occurs in nine habitat types and thrives best, with an ecological optimum in annual vegetation on arable land and annual vegetation in trampled habitats. At present its occurrence is still mostly confined to cities and villages with its invasion being supported by the spread of suburban areas with garden allotments. It spreads, both by seed and stem fragments, colonizing loose well-aerated soils such as garden beds, in glasshouses or heaps of garden substrata. Its ability to grow fast makes it a noxious garden weed, especially early in the season when it profits from competitive superiority over seedlings of other species. It also occurs in joints between sections of pavement, in railway areas and along walls, but only rarely as a weed of agricultural crops.

Oxalis dillenii (*Oxalidaceae*, Fig. 8.18a) is an annual, biennial or short-lived erect, decumbent or prostrate, rarely creeping perennial herbaceous plant that is native to eastern and central North America, where it grows in prairies and broad-leaved forests, but also in a variety of disturbed habitats. It was most likely introduced into Europe during the first half of nineteenth century as a garden ornamental. It became naturalized starting in the second half of the nineteenth century, but its spread was recorded only after World War II. The likely pathway of introduction

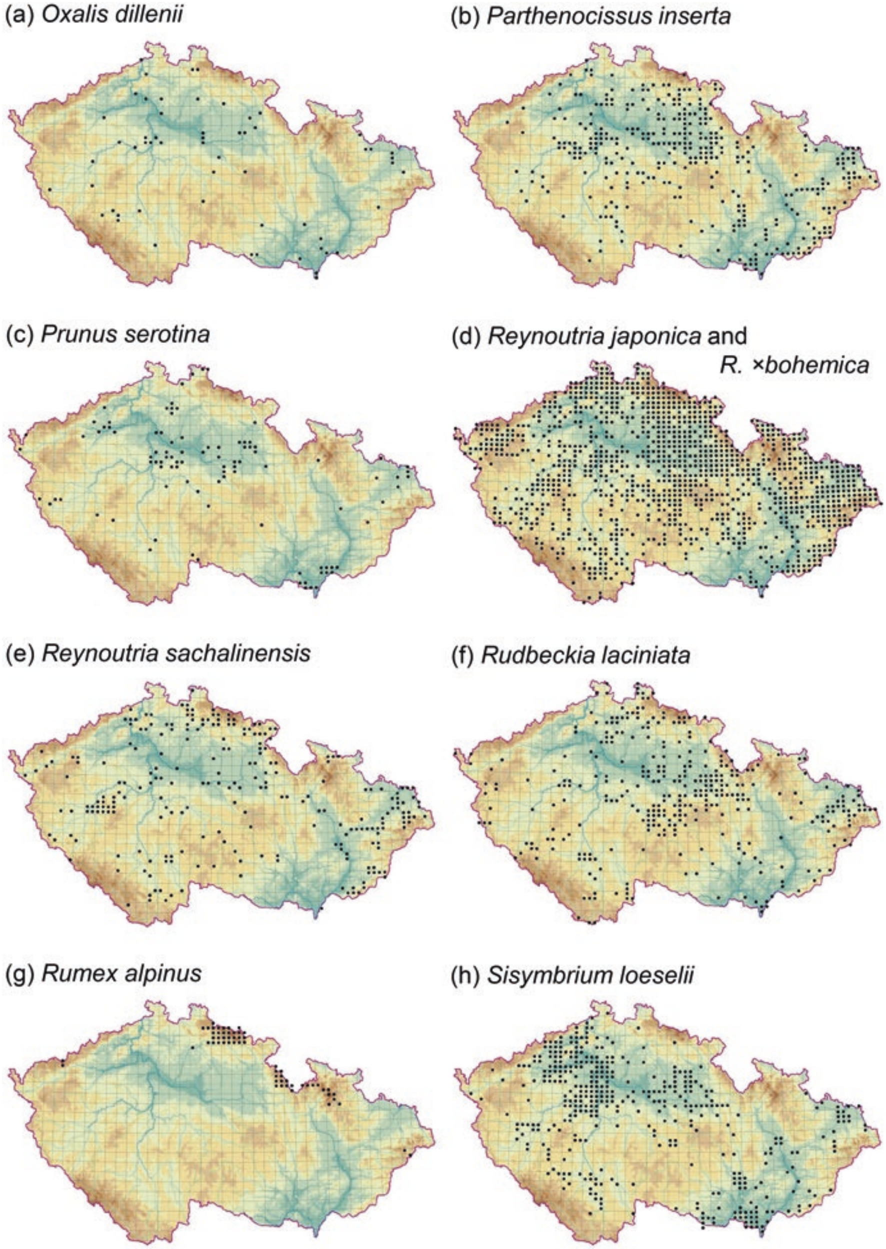

Fig. 8.18 Distribution of selected invasive plant species in the Czech Republic. Note that *Oxalis dillenii* is probably under-recorded

into the Czech Republic is via contaminated soil. It is confined to warm lowland areas and occurs in nine habitats, mostly in settlements and suburban areas. The invasion started during the last decade when this species began to spread in a wide range of habitats such as waste places, rubbish dumps, along paths and railways, field margins, open grassland, root crops and fodder crops. It is likely to continue spreading, but because it is a weak competitor which does not dominate vegetation, it is unlikely to have a serious impact.

Parthenocissus inserta (*Vitaceae*, Fig. 8.18b) is a woody vine native to North America. It was introduced into Europe about 1800 and the first record of an escaped population in the Czech Republic is from 1900. Planted as an ornamental species in parks and gardens, it escapes and spreads in villages and their surroundings, from where it invades alluvial forests in the warmer parts of this country (southern Moravia, central Bohemia). This species requires moist, nutrient-rich soils and spreads slowly, supported by succession of ruderal scrub towards tree-dominated stands. It is able to propagate both by seed and clonally, which enhances its invasive potential and impact on invaded vegetation.

Pinus strobus (*Pinaceae*) is a coniferous tree with its native distribution range in eastern North America and a part of Central America. In its native range it is an important forestry species forming extensive, often monodominant, forests. *Pinus strobus* was introduced into Europe in 1705. The introduction to the Bohemian lands for forestry purposes, with the first record in 1784, predates that for ornamental purposes (1812). In the Czech Republic this species is locally invasive on sandy acidic soils in sandstone areas, especially in the Elbe Sandstones (Bohemian Switzerland), where it gradually outcompetes the native *P. sylvestris*. It was first planted there in 1798 and started to spread in the 1950s in mixed forests, but also in other habitat types, including open vegetation on rock outcrops. Its invasion and impact is restricted to extremely poor soils; the native pine regenerates poorly in *P. strobus*-dominated stands. The invasion in this sandstone area has accelerated since the 1990s and is continuing. This species exerts strong impacts on invaded communities by developing dense stands with a thick layer of slowly decomposing litter.

Populus ×canadensis (*Salicaceae*) is a fast-growing tree up to 40 m tall. It originated as a product of both intentional and spontaneous hybridization of the European *P. nigra* with the North American *P. deltoides* around 1750. This hybrid was widely planted for forestry purposes, in wind-breaks and as an ornamental species in parks and gardens. The first record of cultivation of *P. ×canadensis* in the Czech Republic is from 1852. In the Czech Republic it reproduces both by seed and root suckers and can spread a great distance from plantings. *Populus ×canadensis* invades areas with disturbed soils along streams, which is reflected in its distribution in the Czech Republic being concentrated in floodplains, however, it is often found also in urban and suburban areas and sand pits. It is often planted at sites of former hardwood floodplain forests, but it also spontaneously occurs in more natural alluvial forests, willow galleries on loamy and sandy river banks and in other habitats (14 habitat types, Fig. 8.8). The spread is supported by it being more drought resistant, compared to *P. nigra*. The recorded impact of this hybrid is through further hybridization with the native *P. nigra*, which is at present exacerbated by utilization of *P.* ×*canadensis* as a biofuel plant.

Prunus serotina (*Rosaceae*, Figs. 8.17e and 8.18c) is native to large areas in eastern North America and Central America, where it grows as a tree up to 35 m tall. It reproduces by seed dispersed by birds and small vertebrates, as well as vegetatively through the formation of dense polycorms. Its habitat in its native range is deciduous and pine forests. The species has been introduced into Europe and Asia. The first record of a European introduction is from 1623 in France. *Prunus serotina* was planted as a forest and park tree, and since it is tolerant of air pollution and poor soil, it was also widely used in urban areas, for soil amelioration and for reclamation in mining areas. Following its introduction this species was released from the effect of a parasitic fungus that is known to control its abundance in its native range, which might have contributed to its invasiveness. In the Czech Republic, the species was introduced into cultivation in 1811. In this country it is a small tree or shrub reaching 3–6 m in height. It occurs on various soils, but prefers moist, acidic, well-drained soils. Its niche in the Czech Republic is still rather narrow. It occurs in six habitat types, thriving best in alluvial forests and acidophilous oak forests. It also spreads in forest clearings and along forest paths and roads, reducing the abundance and richness of understorey species. It is also able to establish in abandoned fields on sandy soils and in sand pits. Invasion in the Czech Republic is currently markedly accelerating, especially in the lowlands, being supported by ongoing eutrophication of nutrient-poor forests, and is also beginning to colonize treeless landscapes. The distribution in the country is still rather restricted. *Prunus serotina* exerts an impact on native species by being a strong competitor and forming thickets. Impact on humans can be through bark and seeds that are toxic.

Quercus rubra (*Fagaceae*, Fig. 8.19a) is a tree up to 40 m tall, native to a large area in eastern North America where it is an important source of timber. It grows in a wide range of dry-mesic to mesic sites and occurs in various habitats ranging from nutrient-rich soils to sandy plains and rock outcrops. Currently this species is widely cultivated in the temperate zone of Europe and Asia as a popular forestry and ornamental tree. It was introduced into Europe in 1691 and has been planted in the Czech Republic since 1799. In the Czech Republic, it was mostly used as a garden and park ornamental until several decades ago when it started to be introduced into forest plantations, often in monocultures; in the 2000s it was planted on more than 4000 ha. It is also used for reclaiming post-mining areas and for afforestation of arable land. It is now widespread and spreading mainly in central and eastern Bohemia in the Labe River lowland. It is recorded from 14 habitat types (Fig. 8.8). *Quercus rubra* has a short juvenile period and spreads into surrounding vegetation because it is more shade-tolerant than native oaks. It prefers open forest on light, nutrient-poor soils. Acorns are spread outside forests by birds, but populations are only able to establish in nutrient-poor mesic habitats. As a fast- and vigorously growing, shade-tolerant tree, it has an impact on forest understorey, exacerbated by its slowly decomposing leaf litter inhibiting succession.

Reynoutria japonica* var. *japonica and ***R. sachalinensis*** (*Polygonaceae*, Figs. 8.18d and 8.19b) are rhizomatous perennials native to East Asia, from where

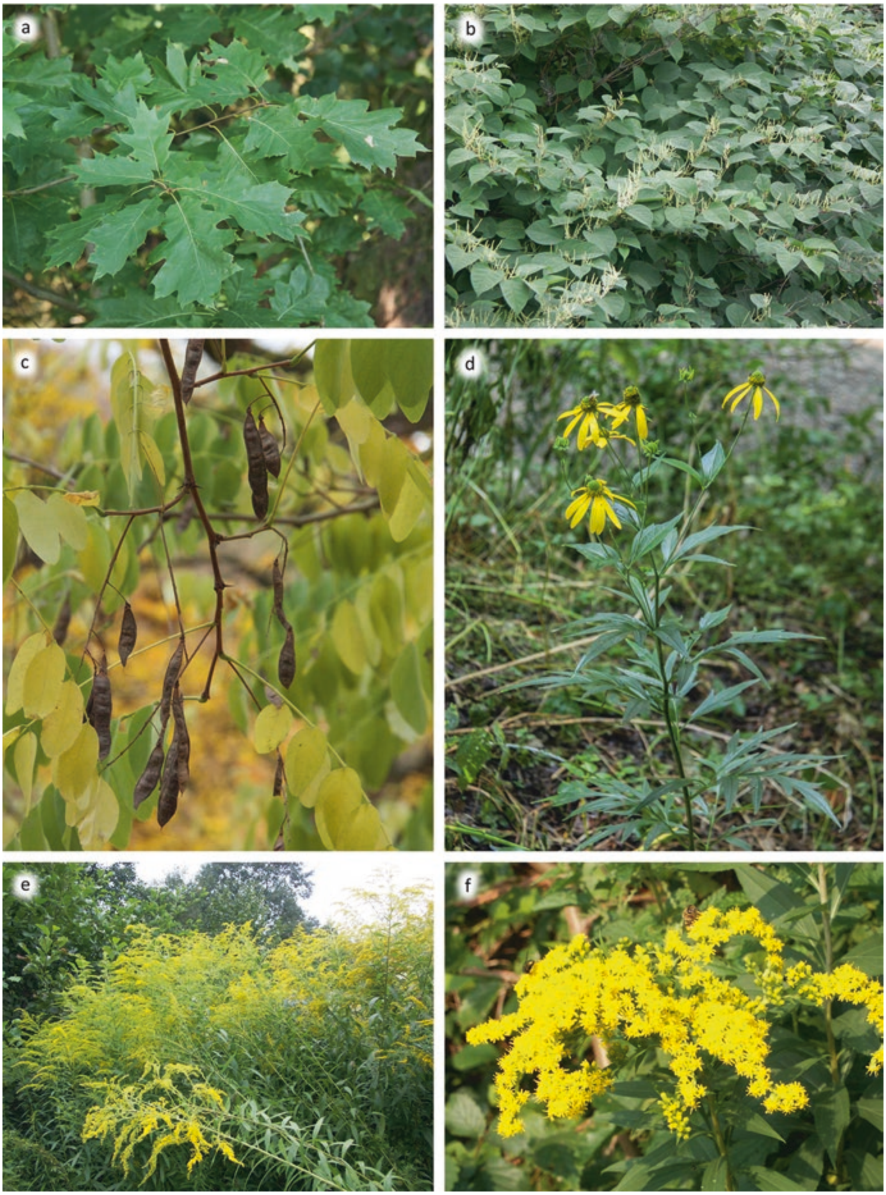

Fig. 8.19 Examples of invasive plant species in the Czech Republic: (**a**) *Quercus rubra*, Chlum u Třeboně, southern Bohemia; (**b**) *Reynoutria japonica*, Prague; (**c**) *Robinia pseudoacacia*, Veverská Bítýška, southern Moravia; (**d**) *Rudbeckia laciniata*, Třeboň, southern Bohemia; (**e**) *Solidago canadensis*, Prague; (**f**) *S. gigantea*, Lednice, southern Moravia. Photo credits: M. Chytrý (a, c, d) and P. Pyšek (b, e, f)

they were introduced into Europe as garden ornamentals and fodder plants in the nineteenth century. In the Czech Republic the genus *Reynoutria* (syn. *Fallopia* p.p.) is represented also by the invasive hybrid ***R. ×bohemica***. This hybrid, first grown in gardens in the UK in 1872, is likely to have arisen on this continent several times independently and is also known from the native range of the parental species. In their native range, species of *Reynoutria* grow along rivers and in disturbed open habitats and at the edges of forests. *Reynoutria japonica* var. *japonica* is also known to colonize volcanic slopes and bare lava fields. The invaded range includes Europe, North America and Australasia. All three taxa have been introduced into the Czech Republic as garden ornamentals; the first record of *R. japonica* var. *japonica* in cultivation is from 1883, and outside cultivation from 1902. *Reynoutria sachalinensis* was first collected in 1921, and the earliest record of the hybrid *R.* ×*bohemica* is from 1950. The invasion occurred in the second half of the twentieth century when this species colonized most of this country. That of the hybrid lagged behind the two parental species but proceeded faster because it is competitively superior to its parents. In the early 2000s, *R. japonica* var. *japonica* was recorded from 1335 localities, *R. sachalinensis* from 261 and the hybrid from 382. Typical of the *Reynoutria* complex of taxa is the large intraspecific variation in ploidy in both the native and invaded ranges, and interspecific hybridization that is common especially in invaded regions. In the Czech Republic, all three taxa are still planted as garden ornamentals. Their dispersal is mainly vegetative by the regeneration from rhizome and stem segments transported in contaminated soil and water. Sexual reproduction rarely occurs, being constrained by the lack of pollen grains in some taxa or inefficient seedling establishment. *Reynoutria japonica* var. *japonica* has been introduced into Europe as a single female clone, which spread across the continent. *Reynoutria* taxa became widely naturalized and are now among the worst invasive plants in the Czech Republic in terms of their impact. They occur in a number of habitat types (12) and are dominant in perennial nitrophilous herbaceous vegetation growing at mesic to wet sites (Fig. 8.8). They are demanding in terms of moisture and nitrogen and invade settlements and floodplains, where the invasion is supported by periodical large-scale disturbances such as floods, during which rhizomes are spread and new habitats suitable for colonization created. The strongest impact is recorded in river floodplains in north-eastern Moravia or northern Bohemia. Due to their high competitive ability, high biomass production and efficient vegetative reproduction, knotweeds are classified as transformer species (sensu Richardson et al. 2000) that cause alterations in water regime and displace native plant species. The invasion by *Reynoutria* taxa has the most severe impact on species richness and diversity of any of the Central European alien plants, reducing the number of species present prior to invasion by 66–86%, depending on the taxon. *Reynoutria* taxa are not reported to affect human health, but do affect the infrastructure by damaging roads and flood-prevention structures, and increasing the erosion potential of rivers. For *R. japonica*, a biological control agent, the psyllid *Aphalara itadori*, was recently released in the UK.

Robinia pseudoacacia (*Fabaceae*, Fig. 8.19c) is a deciduous tree up to 30 m tall. In its native range in central and eastern North America it grows as an early successional species in open and disturbed habitats. The tree has a good regeneration capacity, resprouting well from roots and stumps. The invaded range includes temperate areas. *Robinia pseudoacacia* was introduced into Europe in 1601 as an ornamental species. Later it was used for timber production and for erosion control. It has been cultivated in the Czech Republic since 1710, after which it was widely planted in parks. The first record of its spontaneous occurrence in the wild is from 1874. Since the 1760s it has been planted for timber, erosion control, to support soil eutrophication and as a bee plant. It grows up to ~500 m a.s.l., mostly in central Bohemia and southern Moravia, locally in extensive groves. It is tolerant of drought and air pollution, and grows on sandy, poorly drained and saline substrates. It is resistant to fire and trimming and does not suffer from attacks by pests and diseases. It reproduces by seed, but seedlings only establish following disturbance at warm sites. Spread is by root suckers, and root fragments also contribute to dispersal. This is why most populations are found close to original plantings, whereas long-distance dispersal happens only occasionally, and is mostly linked to transport of substrate from mines, quarries and sand pits. Its local spread is accelerated by attempts at eradication that mostly fail and only serve to stimulate resprouting. The species is recorded in 24 habitats (Fig. 8.8). The whole plant is toxic for humans and cattle, and produces nitrogene compounds and allelopathic substances that inhibit germination and growth of native species. However, stands of *R. pseudoacacia* harbour some scarce native species, especially vernal geophytes (e.g. of the genera *Allium* and *Gagea*), that are able to tolerate the effects of the allelopathic compounds. Its poor ability of spreading over long-distance and the difficulty in eradicating it are probably the reasons why old populations of *R. pseudoacacia* are often tolerated by nature conservation authorities, except when they invade steppe vegetation with a high conservation value.

Rudbeckia laciniata (*Asteraceae*, Figs. 8.18f and 8.19d) is a stout perennial species reproducing both by seed and rhizome fragments. Its native range is in north-eastern Canada and eastern and central USA, where it grows along streams and in wet habitats from the lowlands to the mountains. The invaded range includes Europe, spans central Russia and the Caucasus, China, Japan and New Zealand. It has been planted in Europe since the early seventeenth century and in the Czech Republic since the nineteenth century. At present a number of cultivated forms as well as hybrids with *R. nitida* are frequently planted in gardens, but only the one with the oldest history of planting has escaped from cultivation. The first record in the wild from the Czech Republic dates to 1895. Currently this species is common, namely in eastern, northern and south-western parts of this country. *Rudbeckia laciniata* efficiently spreads by rhizome fragments dispersed by water in riparian habitats, wet meadows, and along roads and railways with disturbed soil. It occurs in nine habitat types. About 80% of the populations persist at invaded sites for many decades, with a maximum of up to 135 years recorded for one clone. This species forms stands that may reach cover values of up to 80–100% and reduce species diversity compared to uninvaded stands by ~30%, depending on the measure used.

Rumex alpinus (*Polygonaceae*, Fig. 8.18g) is a clonal herbaceous perennial up to 1.5 m tall, native to the mountains of Central and Southern Europe, and the Caucasus region and northern Turkey. The species was introduced into mountainous areas elsewhere in Europe, Great Britain, North America and South-east Asia. The first literature record from the Czech Republic is from 1819, but this species might have been introduced as early as the second half of the sixteenth or the early seventeenth century by woodcutters from the Alps coming to the Krkonoše and Orlické hory Mts. This species occurs in mountain ranges in the northern part of this country, with invasive populations concentrated in the Krkonoše Mts where rapid invasion started after World War II, supported by abandonment of mountain grasslands. Plants spread by rhizomes and invade abandoned nutrient-rich meadows or cattle pastures. This species also occurs in a variety of habitats including some semi-natural ones, e.g. gravel riverbanks and fringes of montane brooks. Current spread is supported by the ruderalization of mountain landscapes. Invasive populations may reach a cover of up to 75–100% and reduce species richness of invaded communities by ~50%.

Rumex longifolius* subsp. *sourekii (*Polygonaceae*) is a rhizomatous perennial up to 1.8 m tall. This subspecies is described from the Czech Republic. *Rumex longifolius* is native to the Pyrenees and mountains in Scotland, Ireland and Scandinavia, where it grows in moist grasslands and along streams. The first record of invasive populations in the Czech Republic comes from 1961, and a rapid invasion followed towards the end of the twentieth century; in the 1990s *R. longifolius* subsp. *sourekii* was locally common in the Krkonoše, Krušné hory and Jizerské hory Mts, and it is still spreading. The populations are restricted to mountain areas where they occur at disturbed sites along water courses, in ruderalized or abandoned meadows and pastures, road verges and human settlements, finding their ecological optimum in perennial nitrophilous herbaceous vegetation. The subspecies *sourekii* is much more widespread than the rare type subspecies. Currently it also colonizes suitable habitats at lower altitudes. It is sensitive to mowing. Potential impact of this invasion results from the fact that *Rumex* species easily hybridize; it is recorded that this taxon has produced three hybrids with native species in the Czech Republic.

Sisymbrium loeselii (*Brassicaceae*, Fig. 8.18h) is an annual plant whose native distribution ranges from Southern Europe, including the Mediterranean area, to Central Asia, where it grows on mountain slopes and in disturbed road verges and field edges. The invaded range includes the rest of Europe and North America. The first record in the Czech Republic dates to 1819. It is more common in warm areas. It occurs in a wide range of habitat types (16), being dominant in annual vegetation in ruderal habitats (Fig. 8.8). In the past populations occurred mainly in urban areas, but over the last 30 years it has started to spread into open landscapes. Currently it occurs in villages, quarries and mining areas, on reclaimed spoil heaps and in abandoned fields, and also colonizes disturbed steppe vegetation. It is a competitively weak, early successional thermophilous species colonizing newly created habitats,

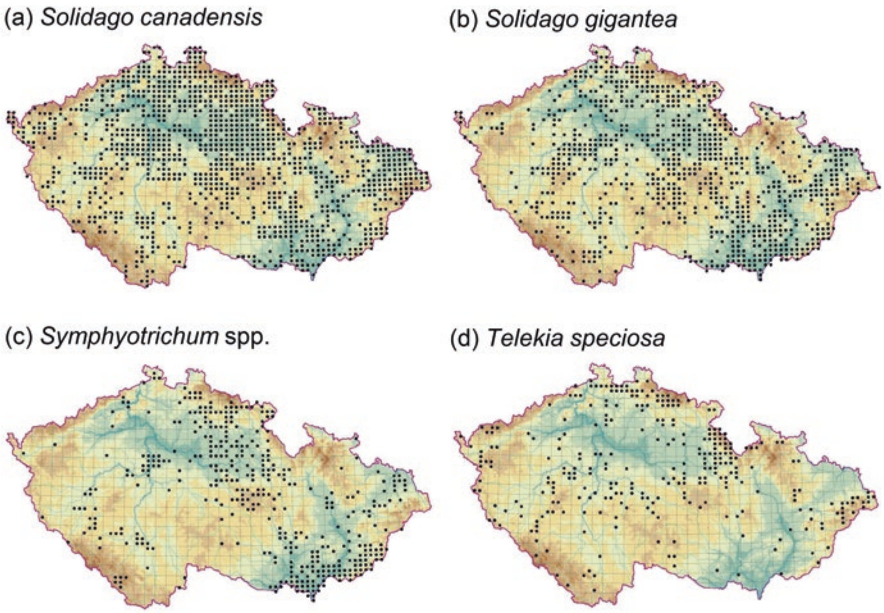

Fig. 8.20 Distribution of selected invasive plant species in the Czech Republic. *Symphyotrichum* spp. includes mainly *S. lanceolatum* and *S. novi-belgii*

which easily becomes established following disturbance in grassland; it rarely occurs on arable land.

Solidago canadensis (*Asteraceae*, Figs. 8.19e and 8.20a) is a rhizomatous herbaceous plant reproducing vegetatively and by seeds. Its native range includes the whole of North America, from Alaska and Labrador in the north to Mexico and Florida in the south. This species is naturalized in Europe, East Asia, Australia and New Zealand. In its native range it occurs at the edges of forests, along rivers and in a variety of disturbed habitats such as abandoned pastures, roadsides, abandoned fields, grasslands and urban areas. It occupies similar habitats in its invaded range. *Solidago canadensis* has been introduced into botanical gardens in Europe as an ornamental plant, with the first record on this continent being from 1645. It was introduced into the Czech Republic as a bee plant and a garden ornamental and first recorded in the wild in 1838. The invasion started as early as the first half of the nineteenth century. It is now common in most parts of this country, occurring in 13 habitat types, being dominant in perennial thermophilous ruderal vegetation (Fig. 8.8). Seeds and rhizomes are often dispersed as soil contaminants. It is suggested that this species supresses co-occurring species by means of allelopathic compounds released into the soil. A negative impact on human health due to pollen allergies is suspected, albeit without supporting data.

Solidago gigantea (*Asteraceae*, Figs. 8.19f and 8.20b) is a rhizomatous perennial plant reproducing both by seeds and vigorous clonal growth of rhizomes. It is native to southern Canada and eastern USA, and has been introduced into Europe, East Asia and New Zealand. In Europe it was first planted as an ornamental plant in botanical gardens. The first record is from 1758 in London. In its native range it is found in grasslands and open forests, while in its invaded range it grows mainly in forests and in disturbed habitats along roads and railways. Introduced as a garden ornamental and important bee plant, it was first reported to occur in the wild in the Czech Republic in 1851. The invasion started in the second half of the nineteenth century, supported by seeds and rhizomes dispersed with contaminated soil. By the 1930s it formed stands along rivers and also started to spread into disturbed sites such as coal mining slag heaps in north-eastern Moravia. Compared to its congener *S. canadensis*, this species forms denser stands, prefers moister and more nutrient-rich soils and is less common. It is also more confined to riverbanks and floodplains of large rivers. It occurs in 13 habitat types and is dominant in perennial nitrophilous herbaceous vegetation at mesic and wet sites (Fig. 8.8). Impact of this species on native vegetation is similar to that of *S. canadensis*. It is known to reduce the species richness and diversity of invaded plant communities by ~25–30%.

Symphoricarpos albus (*Caprifoliaceae*, Fig. 8.21a) is a shrub with a height of up to 2 m, native to the western part of North America. This species has been introduced outside its native range, including the Czech Republic, as an ornamental commonly planted in parks and gardens, along fences and roads, and in forests as shelters for pheasants. Its introduced range is almost cosmopolitan. The earliest record from Europe is probably from 1879. In the Czech Republic, it only occasionally reproduces by seed, relying on vegetative dispersal, and occurs scattered throughout the whole country, most frequently being planted and subsequently escaping in colline to submontane areas. Besides scrub vegetation it is also found in perennial nitrophilous herbaceous vegetation at mesic sites. It occurs in nine habitat types, including semi-natural vegetation such as riverine reed stands, fringes of montane brooks, willow galleries on river banks and alluvial forests. Populations persist at invaded sites for a long time and, to some extent, are resistant to shading by trees; very few other species co-occur with it in the understorey.

Symphyotrichum lanceolatum (syn. *Aster lanceolatus*; *Asteraceae*, Figs. 8.20c and 8.21b) is a herbaceous perennial native to eastern North America. It was introduced into Europe in 1837 and became naturalized across most of the continent, from south-western France to southernmost Scandinavia, up to central Russia and with isolated occurrences on the Iberian Peninsula. It is frequently planted in the Czech Republic, escaping from cultivation and invading a wide range of habitat types (19), including semi-natural riparian habitats (Fig. 8.8). It occurs as a dominant in perennial nitrophilous herbaceous vegetation at mesic sites and in nitrophilous herbaceous fringes of lowland rivers, and is often found thriving in reed- and tall-sedge beds. The invasion is ongoing, particularly on river banks in southern Moravia, but also along rivers in northern Bohemia. It is known to exert significant impacts on invaded communities. Occasional mowing of *Symphyotrichum* spp.

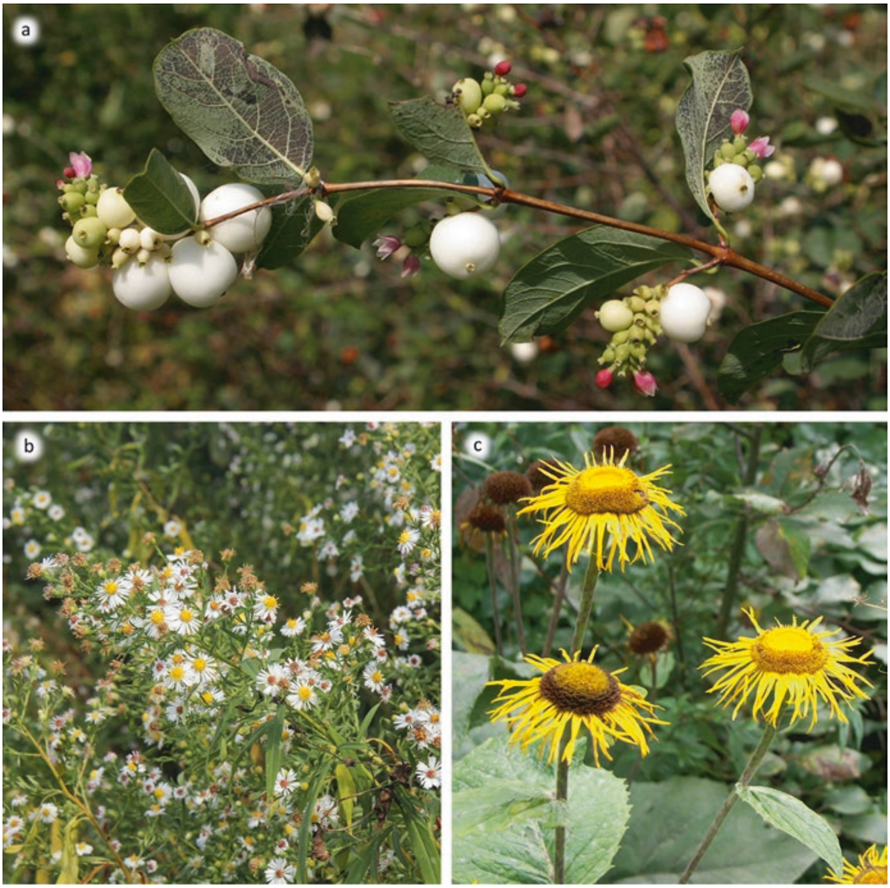

Fig. 8.21 Examples of invasive plant species in the Czech Republic: (**a**) *Symphoricarpos albus*, Žďár nad Sázavou, Bohemian-Moravian Highlands; (**b**) *Symphyotrichum lanceolatum*, Mikulov, southern Moravia; (**c**) *Telekia speciosa*, Průhonice, central Bohemia. Photo credits: M. Chytrý (a, b) and P. Pyšek (c)

stands (as for *Helianthus*, *Rumex* and *Solidago* mentioned above) causes their rapid regeneration resulting in emergence of more dense stands.

Symphyotrichum novi-belgii (syn. *Aster novi-belgii*; *Asteraceae*, Fig. 8.20c) is a perennial species with a native distribution range covering a ~150-km-wide belt along the Atlantic coast of North America, from the Appalachian Mts to southern Canada. As for the previous species, natural habitats include riparian communities along rivers and lakes. It is naturalized in Europe, from northern Italy to southern Scandinavia, in the UK and France, with isolated occurrences in Romania and the central part of European Russia. It was introduced into Europe in 1710. The first record from the Czech Republic is from 1850. Planted as an ornamental, this species is less invasive than *S. lanceolatum*, occurring in 12 habitat types, but it also invades semi-natural habitats such as alluvial meadows of lowland rivers. It occurs as a dominant in perennial nitrophilous herbaceous vegetation at mesic and wet sites. It is known to reduce the species diversity of invaded communities by ~30–40%, depending on the measure and scale.

Symphyotrichum* ×*salignum (syn. *Aster* ×*salignus*; *Asteraceae*, Fig. 8.20c) is an anecophyte resulting from the hybridization of the above two North American species, *S. lanceolatus* and *S. novi-belgii*, which most likely happened in a European garden. It is almost sterile, with less than 0.1% of cypselae ripening. At present it is cultivated and naturalized all over Europe and was first collected in the wild in 1872. The distribution in the Czech Republic is scattered. It invades mainly riparian scrub.

Symphyotrichum versicolor (syn. *Aster* ×*versicolor*; *Asteraceae*, Fig. 8.20c) is considered to be a product of an artificial crossing between *S. laevis* and *S. novi-belgii*, probably in Europe. It is planted as an ornamental in most countries in Central and Western Europe, where it has escaped from cultivation and has become naturalized. In the Czech Republic it occurs in six habitat types, invading mainly perennial thermophilous ruderal vegetation and perennial nitrophilous herbaceous vegetation at mesic and wet sites.

Telekia speciosa (*Asteraceae*, Figs. 8.20d and 8.21c) is a rhizomatous perennial up to 2 m tall, native to the mountains of Southern and Eastern Europe, northern Anatolia and the Caucasus. *Telekia speciosa* grows at the edges of mountain forests or clearings and along roads and brooks. It is naturalized in areas outside its native distribution in Europe and the European part of Russia, in similar habitats as in its native range. This nitrophilous, shade-tolerant species, requiring moist clayey soils, was introduced as a garden ornamental into the Czech Republic, with the first record in the wild dated around 1820. It is still commonly planted and escapes along streams, in the surroundings of parks and gardens where it is cultivated, as well as in other habitat types such as forest margins, old forest clearings or unmown road ditches. It thrives best in perennial nitrophilous herbaceous vegetation at mesic and wet sites. It occurs most commonly at middle altitudes, especially in north-eastern Bohemia. Impact on human health is documented: some people are allergic to its pollen and it can also cause skin irritation.

Acknowledgements We thank Jiří Danihelka, Zdeněk Kaplan and Marinus Werger for helpful comments on the manuscript, Ondřej Hájek for preparing the maps, Zdeňka Lososová for a photograph, Kryštof Chytrý for preparing photographic plates, Pavel Dřevojan and Zuzana Sixtová for editing the list of references, Tony Dixon for improving our English and Irena Axmanová for her help with proof correcting. This work was supported by project no. 14-36079G, Centre of Excellence Pladias (Czech Science Foundation), and long-term research development project no. RVO 67985939 (The Czech Academy of Sciences).

References

Aikio S, Duncan RP, Hulme PE (2010) Lag-phases in alien plant invasions: separating the facts from the artefacts. Oikos 119:370–378

Alexander JM, Kuffer C, Daehler CC, Edwards PJ, Pauchard A, Seipel T (2011) Assembly of nonnative floras along elevational gradients explained by directional ecological filtering. Proceedings of the National Academy of Sciences of the USA 108:656–661

Balogh L, Dancza I, Király G (2004) A magyarországi neofitonok időszerű jegyzéke, és besorolásuk inváziós szempontból [A recent list of neophytes in Hungary and their classification according to their success]. In: Mihály B, Botta-Dukát Z (eds) Biológiai inváziók Magyarországon: Özönnövények [Biological invasions in Hungary: invasive plants]. Természet BÚVÁR Alapítvány Kiadó, Budapest, pp 61–92

Bastl M, Kočár P, Prach K, Pyšek P (1997) The effect of successional age and disturbance on the establishment of alien plants in man-made sites: an experimental approach. In: Brock JH, Wade M, Pyšek P, Green D (eds) Plant invasions: studies from North America and Europe. Backhuys Publishers, Leiden, pp 191–201

Becker T, Dietz H, Billeter R, Bush MB, Buschmann H, Edwards PJ (2005) Altitudinal distribution of alien plant species in the Swiss Alps. Perspectives in Plant Ecology, Evolution and Systematics 7:173–183

Blackburn TM, Pyšek P, Bacher S, Carlton JT, Duncan RP, Jarošík V, Wilson JRU, Richardson DM (2011) A proposed unified framework for biological invasions. Trends in Ecology and Evolution 26:333–339

Blackburn TM, Essl F, Evans T, Hulme PE, Jeschke JM, Kühn I, Kumschick S, Marková Z, Mrugała A, ... Bacher S (2014) A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biology 12:e1001850

Celesti-Grapow L, Alessandrini A, Arrigoni PV, Banfi E, Bernardo L, Bovio M, Brundu G, Cagiotti MR, Camarda I, ... Blasi C (2009) Inventory of the non-native flora of Italy. Plant Biosystems 143:386–430

Chytrý M (ed) (2007–2013) Vegetace České republiky 1–4 [Vegetation of the Czech Republic 1–4]. Academia, Praha

Chytrý M, Rafajová M (2003) Czech national phytosociological database: basic statistics of the available vegetation-plot data. Preslia 75:1–15

Chytrý M, Pyšek P, Tichý L, Knollová I, Danihelka J (2005) Invasions by alien plants in the Czech Republic: a quantitative assessment across habitats. Preslia 77:339–354

Chytrý M, Jarošík V, Pyšek P, Hájek O, Knollová I, Tichý L, Danihelka J (2008a) Separating habitat invasibility by alien plants from the actual level of invasion. Ecology 89:1541–1553

Chytrý M, Maskell LC, Pino J, Pyšek P, Vilà M, Font X, Smart SM (2008b) Habitat invasions by alien plants: a quantitative comparison among Mediterranean, subcontinental and oceanic regions of Europe. Journal of Applied Ecology 45:448–458

Chytrý M, Pyšek P, Wild J, Pino J, Maskell LC, Vilà M (2009a) European map of alien plant invasions, based on the quantitative assessment across habitats. Diversity and Distributions 15:98–107

Chytrý M, Wild J, Pyšek P, Tichý L, Danihelka J, Knollová I (2009b) Maps of the level of invasion of the Czech Republic by alien plants. Preslia 81:187–207

Chytrý M, Wild J, Pyšek P, Jarošík V, Dendoncker N, Reginster I, Pino J, Maskell L, Vilà M, ... Settele J (2012) Projecting trends in plant invasions in Europe under different scenarios of future land-use change. Global Ecology and Biogeography 21:75–87

Čuda J, Skálová H, Janovský Z, Pyšek P (2015) Competition among native and invasive *Impatiens* species: the roles of environmental factors, population density and life stage. AoB Plants 7:plv033

Čuda J, Skálová H, Janovský Z, Pyšek P (2016) Juvenile biological traits of *Impatiens* species are more strongly associated with naturalization in temperate climate than their adult traits. Perspectives in Plant Ecology, Evolution and Systematics 20:1–10

Danihelka J (2013) Botanické součty, rozdíly a podíly [Botanical counts]. Živa 61:69–72
Danihelka J, Chrtek J Jr, Kaplan Z (2012) Checklist of vascular plants of the Czech Republic. Preslia 84:647–811
Davis MA, Grime JP, Thompson K (2000) Fluctuating resources in plant communities: a general theory of invasibility. Journal of Ecology 88:528–534
di Castri F (1989) History of biological invasions with special emphasis on the Old World. In: Drake JA, Mooney HA, di Castri F, Groves RH, Kruger FJ, Rejmánek M, Williamson M (eds) Biological invasions: a global perspective. Wiley, Chichester, pp 1–30
Dostál P, Dawson W, van Kleunen M, Keser LH, Fischer M (2013) Central European plant species from more productive habitats are more invasive at a global scale. Global Ecology and Biogeography 22:64–72
Dostálek J, Frantík T, Šilarová V (2016) Changes in the distribution of alien plants along roadsides in relation to adjacent land use over the course of 40 years. Plant Biosystems 150:442–448
Essl F, Rabitsch W (eds) (2002) Neobiota in Österreich. Umweltbundesamt, Wien
Essl F, Dullinger D, Kleinbauer I (2009) Changes in the spatio-temporal patterns and habitat preferences of *Ambrosia artemisiifolia* during the invasion of Austria. Preslia 81:119–133
Essl F, Dullinger S, Rabitsch W, Hulme PE, Hülber K, Jarošík V, Kleinbauer I, Krausmann F, Kühn I, ... Pyšek P (2011) Socioeconomic legacy yields an invasion debt. Proceedings of the National Academy of Sciences of the USA 108:203–207
European Union (2014) Regulation (EU) No 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species. Official Journal of the European Union L 317:35–55
Genovesi P, Carboneras C, Vilà M, Walton P (2014) EU adopts innovative legislation on invasive species: a step towards a global response to biological invasions? Biological Invasions 17:1307–1311
Hejda M (2013) Do species differ in their ability to coexist with the dominant alien *Lupinus polyphyllus*? A comparison between two distinct invaded ranges and a native range. NeoBiota 17:39–55
Hejda M, Pyšek P, Jarošík V (2009a) Impact of invasive plants on the species richness, diversity and composition of invaded communities. Journal of Ecology 97:393–403
Hejda M, Pyšek P, Pergl J, Sádlo J, Chytrý M, Jarošík V (2009b) Invasion success of alien plants: do habitat affinities in the native distribution range matter? Global Ecology and Biogeography 18:372–382
Hejda M, Chytrý M, Pergl J, Pyšek P (2015) Native-range habitats of invasive plants: are they similar to invaded-range habitats and do they differ according to the geographical direction of invasion? Diversity and Distributions 21:312–321
Hejda M, Štajerová K, Pyšek P (2017) Dominance has a biogeographical component: do plants tend to exert stronger impacts in their invaded rather than native range? Journal of Biogeography 44:18–27
Hejný S, Jehlík V, Kopecký K, Kropáč Z, Lhotská M (1973) Karanténní plevele Československa [Quarantine weeds of Czechoslovakia]. Studie ČSAV 1973(8):1–156
Hejný S, Kopecký K, Jehlík V, Krippelová T (1979) Přehled ruderálních rostlinných společenstev Československa [Survey of ruderal plant communities of Czechoslovakia]. Rozpravy Československé akademie věd, Řada matematických a přírodních věd 89(2):1–100
Hejný S, Slavík B, Chrtek J, Tomšovic P, Kovanda M (eds) (1988) Květena České socialistické republiky 1 [Flora of the Czech Socialist Republic 1]. Academia, Praha
Hejný S, Slavík B, Hrouda L, Skalický V (eds) (1990) Květena České republiky 2 [Flora of the Czech Republic 2]. Academia, Praha
Hejný S, Slavík B, Kirschner J, Křísa B (eds) (1992) Květena České republiky 3 [Flora of the Czech Republic 3]. Academia, Praha
Holub J, Jirásek V (1967) Zur Vereinheitlichung der Terminologie in der Phytogeographie. Folia Geobotanica et Phytotaxonomica 2:69–113
Horáčková J, Juřičková L, Jarošík V, Šizling A, Pyšek P (2014) Invasiveness does not predict impact: response of native land snail communities to plant invasions in riparian habitats. PLoS One 9:e108296

Hulme PE, Pyšek P, Nentwig W, Vilà M (2009) Will threat of biological invasions unite the European Union? Science 324:40–41

Jandová K, Klinerová T, Müllerová J, Pyšek P, Pergl J, Cajthaml T, Dostál P (2014) Long-term impact of *Heracleum mantegazzianum* invasion on soil chemical and biological characteristics. Soil Biology and Biochemistry 68:270–278

Jehlík V (ed) (1998) Cizí expanzivní plevele České republiky a Slovenské republiky [Alien expansive weeds of the Czech Republic and the Slovak Republic]. Academia, Praha

Jehlík V, Hejný S (1974) Main migration routes of adventitious plants in Czechoslovakia. Folia Geobotanica et Phytotaxonomica 9:241–248

Kalníková V (2012) Rozšíření invazních neofytů a sukcese na štěrkových náplavech na tocích Moravskoslezských Beskyd a jejich podhůří [Distribution of invasive neophytes and succession on gravel sediments along streams of the Moravskoslezské Beskydy Mts and their foothills]. MSc thesis, Masaryk University, Brno

Kalusová V (2009) Rostlinné invaze v aluviálních biotopech dolního toku Moravy a Dyje [Plant invasions in floodplain habitats of the lower Morava nad Dyje Rivers]. MSc thesis, Masaryk University, Brno

Kalusová V, Chytrý M, Kartesz JT, Nishino M, Pyšek P (2013) Where do they come from and where do they go? European habitats as donors of alien plants globally. Diversity and Distributions 19:199–214

Kalusová V, Chytrý M, Peet RK, Wentworth TR (2014) Alien species pool influences the level of habitat invasion in intercontinental exchange of alien plants. Global Ecology and Biogeography 23:1366–1375

Kalusová V, Chytrý M, Peet RK, Wentworth TR (2015) Intercontinental comparison of habitat levels of invasion between temperate North America and Europe. Ecology 96:3363–3373

Kaplan Z (2012) Flora and phytogeography of the Czech Republic. Preslia 84:505–574

Kowarik I (1995) Time lags in biological invasions with regard to the success and failure of alien species. In: Pyšek P, Prach K, Rejmánek M, Wade M (eds) Plant invasions: general aspects and special problems. SPB Academic Publishers, Amsterdam, pp 15–38

Křivánek M (2006) Biologické invaze a možnosti jejich předpovědi [Biological invasions and possibilities for their prediction]. Acta Průhoniciana 84:5–73

Křivánek M, Pyšek P (2006) Predicting invasions by woody species in a temperate zone: a test of three risk assessment schemes in the Czech Republic (Central Europe). Diversity and Distributions 12:319–327

Kubešová M, Moravcová L, Suda J, Jarošík V, Pyšek P (2010) Naturalized plants have smaller genomes than their non-invading relatives: a flow cytometric analysis of the Czech alien flora. Preslia 82:81–96

Kumschick S, Bacher S, Evans T, Marková Z, Pergl J, Pyšek P, Vaes-Petignat S, van der Veer G, Vilà M, Nentwig W (2015) Comparing impacts of alien plants and animals using a standard scoring system. Journal of Applied Ecology 52:552–561

Lambdon PW, Pyšek P, Basnou C, Hejda M, Arianoutsou M, Essl F, Jarošík V, Pergl J, Winter M, … Hulme PE (2008) Alien flora of Europe: species diversity, temporal trends, geographical patterns and research needs. Preslia 80:101–149

Linc O (2012) Efektivita likvidace invazních druhů v České republice na příkladu bolševníku velkolepého [The efficiency of eradication of invasive species in the Czech Republic exemplified by *Heracleum mantegazzianum*]. BSc thesis, University of Economics, Prague

Lockwood JL, Cassey P, Blackburn TM (2009) The more you introduce the more you get: the role of colonization pressure and propagule pressure in invasion ecology. Diversity and Distributions 15:904–910

Lonsdale WM (1999) Global patterns of plant invasions and the concept of invasibility. Ecology 80:1522–1536

Lososová Z, Cimalová Š (2009) Effects of different cultivation types on native and alien weed species richness and diversity in Moravia (Czech Republic). Basic and Applied Ecology 10:456–465

Lososová Z, Grulich V (2009) Chorological spectra of arable weed vegetation types in the Czech Republic. Phytocoenologia 39:235–252
Lososová Z, Simonová D (2008) Changes during the 20th century in species composition of synanthropic vegetation in Moravia (Czech Republic). Preslia 80:291–305
Lososová Z, Chytrý M, Cimalová Š, Kropáč Z, Otýpková Z, Pyšek P, Tichý L (2004) Weed vegetation of arable land in Central Europe: gradients of diversity and species composition. Journal of Vegetation Science 15:415–422
Lososová Z, de Bello F, Chytrý M, Kühn I, Pyšek P, Sádlo J, Winter M, Zelený D (2015) Alien plants invade more phylogenetically clustered communities and cause their even stronger clustering. Global Ecology and Biogeography 24:786–794
Matějček T (2008) Výskyt invazních druhů rostlin v břehové vegetaci vybraných vodních toků [Occurrence of invasive plant species in riverbank vegetation along selected water courses]. Zprávy České botanické společnosti 43, Materiály 23:169–182
McDougall KL, Alexander JM, Haider S, Pauchard A, Walsh NG, Kueffer C (2011) Alien flora of mountains: global comparisons for the development of local preventive measures against plant invasions. Diversity and Distributions 17:103–111
Medvecká J, Kliment J, Májeková J, Halada Ľ, Zaliberová M, Gojdičová E, Feráková V, Jarolímek I (2012) Inventory of the alien flora of Slovakia. Preslia 84:257–310
Mihulka S (1998) The effect of altitude on the pattern of plant invasions: a field test. In: Starfinger U, Edwards K, Kowarik I, Williamson M (eds) Plant invasions: ecological mechanisms and human responses. Backhuys, Leiden, pp 313–320
Moravcová L, Pyšek P, Jarošík V, Havlíčková V, Zákravský P (2010) Reproductive characteristics of neophytes in the Czech Republic: traits of invasive and non-invasive species. Preslia 82:365–390
Müllerová J, Pyšek P, Jarošík V, Pergl J (2005) Aerial photographs as a tool for assessing the regional dynamics of the invasive plant species *Heracleum mantegazzianum*. Journal of Applied Ecology 42:1042–1053
Nentwig W, Bacher S, Pyšek P, Vilà M, Kumschick S (2016) The Generic Impact Scoring System (GISS): a standardized tool to quantify the impacts of alien species. Environmental Monitoring and Assessment 188:315
Niklfeld H (1998) Mapping the flora of Austria and the eastern Alps. Revue Valdôtaine d'Histoire Naturelle, Suppl 51:53–62
Ööpik M, Kukk T, Kull K, Kull T (2008) The importance of human mediation in species establishment: analysis of the alien flora of Estonia. Boreal Environment Research 13:53–67
Otýpková Z, Chytrý M, Tichý L, Pechanec V, Jongepier JW, Hájek O (2011) Floristic diversity patterns in the White Carpathians Biosphere Reserve, Czech Republic. Biologia 66:266–274
Pergl J, Dušek J, Hošek M, Knapp M, Simon O, Berchová K, Bogdan V, Černá M, Poláková S, … Svobodová J (2016) Metodiky mapování a monitoringu invazních (vybraných nepůvodních) druhů [Guidelines for mapping and monitoring invasive (selected alien) species]. Botanický ústav AV ČR, Průhonice
Pergl J, Genovesi P, Pyšek P (2016b) Better management of alien species. Nature 531:173
Pergl J, Sádlo J, Petřík P, Danihelka J, Chrtek J Jr, Hejda M, Moravcová L, Perglová I, Štajerová K, Pyšek P (2016c) Dark side of the fence: ornamental plants as a source for spontaneous flora of the Czech Republic. Preslia 88:163–184
Pergl J, Sádlo J, Petrusek A, Laštůvka Z, Musil J, Perglová I, Šanda R, Šefrová H, Šíma J, … Pyšek P (2016d) Black, Grey and Watch Lists of alien species in the Czech Republic based on environmental impacts and management strategy. NeoBiota 28:1–37
Prach K, Řehounková K, Konvalinková P, Trnková R (2008) Invaze a sukcese [Invasion and succession]. Zprávy České botanické společnosti 43, Materiály 23:41–49
Pyšek P, Chytrý M (2014) Habitat invasion research: where vegetation science and invasion ecology meet. Journal of Vegetation Science 25:1181–1187
Pyšek P, Jarošík V (2005) Residence time determines the distribution of alien plants. In: Inderjit (ed) Invasive plants: ecological and agricultural aspects. Birkhäuser Verlag, Basel, pp 77–96

Pyšek P, Jarošík V, Kučera T (2002a) Patterns of invasion in temperate nature reserves. Biological Conservation 104:13–24
Pyšek P, Sádlo J, Mandák B (2002b) Catalogue of alien plants of the Czech Republic. Preslia 74:97–186
Pyšek P, Jarošík V, Kučera T (2003) Inclusion of native and alien species in temperate nature reserves: an historical study from Central Europe. Conservation Biology 17:1414–1424
Pyšek P, Sádlo J, Mandák B, Jarošík V (2003b) Czech alien flora and a historical pattern of its formation: what came first to Central Europe? Oecologia 135:122–130
Pyšek P, Chocholoušková Z, Pyšek A, Jarošík V, Chytrý M, Tichý L (2004) Trends in species diversity and composition of urban vegetation over three decades. Journal of Vegetation Science 15:781–788
Pyšek P, Richardson DM, Rejmánek M, Webster G, Williamson M, Kirschner J (2004b) Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. Taxon 53:131–143
Pyšek P, Jarošík V, Chytrý M, Kropáč Z, Tichý L, Wild J (2005) Alien plants in temperate weed communities: prehistoric and recent invaders occupy different habitats. Ecology 86:772–785
Pyšek P, Křivánek M, Jarošík V (2009) Planting intensity, residence time, and species traits determine invasion success of alien woody species. Ecology 90:2734–2744
Pyšek P, Jarošík V, Hulme PE, Kühn I, Wild J, Arianoutsou M, Bacher S, Chiron F, Didžiulis V, ... Winter M (2010) Disentangling the role of environmental and human pressures on biological invasions across Europe. Proceedings of the National Academy of Sciences of the USA 107:12157–12162
Pyšek P, Jarošík V, Chytrý M, Danihelka J, Kühn I, Pergl J, Tichý L, Biesmeijer J, Ellis WN, ... Settele J (2011a) Successful invaders co-opt pollinators of native flora and accumulate insect pollinators with increasing residence time. Ecological Monographs 81:277–293
Pyšek P, Jarošík V, Pergl J (2011b) Alien plants introduced by different pathways differ in invasion success: unintentional introductions as greater threat to natural areas? PLoS One 6:e24890
Pyšek P, Jarošík V, Pergl J, Wild J (2011c) Colonization of high altitudes by alien plants over the last two centuries. Proceedings of the National Academy of Sciences of the USA 108:439–440
Pyšek P, Chytrý M, Pergl J, Sádlo J, Wild J (2012a) Plant invasions in the Czech Republic: current state, introduction dynamics, invasive species and invaded habitats. Preslia 84:575–630
Pyšek P, Danihelka J, Sádlo J, Chrtek J Jr, Chytrý M, Jarošík V, Kaplan Z, Krahulec F, Moravcová L, … Tichý L (2012b) Catalogue of alien plants of the Czech Republic (2nd edition): checklist update, taxonomic diversity and invasion patterns. Preslia 84:155–255
Pyšek P, Jarošík V, Hulme PE, Pergl J, Hejda M, Schaffner U, Vilà M (2012) A global assessment of invasive plant impacts on resident species, communities and ecosystems: the interaction of impact measures, invading species' traits and environment. Global Ecology and Biogeography 18:1725–1737
Řehounková K, Prach K (2008) Spontaneous vegetation succession in gravel-sand pits: a potential for restoration. Restoration Ecology 16:305–312
Reif J, Hanzelka J, Kadlec T, Štrobl M, Hejda M (2016) Conservation implications of cascading effects among groups of organisms: the alien tree *Robinia pseudoacacia* in the Czech Republic as a case study. Biological Conservation 198:50–59
Rejmánek M (1989) Invasibility of plant communities. In: Drake JA, Mooney HA, di Castri F, Groves RH, Kruger FJ, Rejmánek M, Williamson M (eds) Biological invasions: a global perspective. Wiley, Chichester, pp 369–388
Rejmánek M, Richardson DM, Pyšek P (2005) Plant invasions and invasibility of plant communities. In: van der Maarel E (ed) Vegetation ecology. Blackwell, Oxford, pp 332–355
Richardson DM, Pyšek P (2012) Naturalization of introduced plants: ecological drivers of biogeographic patterns. New Phytologist 196:383–396
Richardson DM, Pyšek P, Rejmánek M, Barbour MG, Panetta FD, West CJ (2000) Naturalization and invasion of alien plants: concepts and definitions. Diversity and Distributions 6:93–107

Roy HE, Adriaens T, Aldridge DC, Bacher S, Bishop JDD, Blackburn TM, Branquart E, Brodie J, Carboneras C, … Zenetos A (2015) Invasive alien species – prioritising prevention efforts through horizon scanning. European Commission, Brussels

Rumlerová Z, Vilà M, Pergl J, Nentwig W, Pyšek P (2016) Scoring environmental and socioeconomic impacts of alien plants invasive in Europe. Biological Invasions 18:3697–3711

Sádlo J, Chytrý M, Pyšek P (2007) Regional species pools of vascular plants in habitats of the Czech Republic. Preslia 79:303–321

Schönfelder P (1999) Mapping the flora of Germany. Acta Botanica Fennica 162:43–53

Šenová V, Matějček T (2013) Zatížení biotopů geograficky nepůvodními druhy rostlin na příkladu povodí Ploučnice [Level of invasion by alien plants across habitats in the basin of the Ploučnice River]. Geografie 118:356–371

Šíma J (2008) Právní úprava problematiky nepůvodních druhů rostlin v České republice a ve světě [Legislation related to non-native species in the Czech Republic and the world]. Zprávy České botanické společnosti 43, Materiály 23:213–218

Simonová D, Lososová Z (2008) Which factors determine plant invasions in man-made habitats in the Czech Republic? Perspectives in Plant Ecology, Evolution and Systematics 10:89–100

Skálová H, Guo W-Y, Wild J, Pyšek P (2017) *Ambrosia artemisiifolia* in the Czech Republic: historic and current invasion pathways predict future spread. Preslia 89:1–16

Slavík B, Smejkal M, Dvořáková M, Grulich V (eds) (1995) Květena České republiky 4 [Flora of the Czech Republic 4]. Academia, Praha

Slavík B, Chrtek J Jr, Tomšovic P (eds) (1997) Květena České republiky 5 [Flora of the Czech Republic 5]. Academia, Praha

Slavík B, Chrtek J Jr, Štěpánková J (eds) (2000) Květena České republiky 6 [Flora of the Czech Republic 6]. Academia, Praha

Slavík B, Štěpánková J, Štěpánek J (eds) (2004) Květena České republiky 7 [Flora of the Czech Republic 7]. Academia, Praha

Sojneková M, Chytrý M (2015) From arable land to species-rich semi-natural grasslands: succession in abandoned fields in a dry region of central Europe. Ecological Engineering 77:373–381

Štěpánková J, Chrtek J Jr, Kaplan Z (eds) (2010) Květena České republiky 8 [Flora of the Czech Republic 8]. Academia, Praha

Strayer DL, D'Antonio C, Essl F, Fowler M, Geist J, Hilt S, Jarić I, Jöhnk K, Jones CG, ... Jeschke JM (2017) Boom-bust dynamics in biological invasions: definitions, causes, detection, and description. Ecology Letters 20:1337–1350

Tokarska-Guzik B (2005) The establishment and spread of alien plant species (kenophytes) in the flora of Poland. Wydawnictwo Uniwersytetu Śląskiego, Katowice

van Kleunen M, Dawson W, Essl F, Pergl J, Winter M, Weber E, Kreft H, Weigelt P, Kartesz J … Pyšek P (2015) Global exchange and accumulation of non-native plants. Nature 525:100–103

Verloove F (2006) Catalogue of neophytes in Belgium (1800–2005). Scripta Botanica Belgica 39:1–89

Vilà M, Pujadas J (2001) Land-use and socio-economic correlates of plant invasions in European and North African countries. Biological Conservation 100:397–401

Vilà M, Basnou C, Pyšek P, Josefsson M, Genovesi P, Gollasch S, Nentwig W, Olenin S, Roques A, ... DAISIE partners (2010) How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. Frontiers in Ecology and the Environment 8:135–144

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

Vítková M, Müllerová J, Sádlo J, Pergl J, Pyšek P (2017) Black locust (*Robinia pseudoacacia*) beloved and despised: a story of an invasive tree. Forest Ecology and Management 384:287–302

Vymyslický T (2001) Rozšíření vybraných invazních druhů rostlin na aluviích jihomoravských řek [Distribution of selected invasive plant species in the floodplains of the southern Moravian rivers]. MSc thesis, Masaryk University, Brno

Williamson M (1996) Biological invasions. Chapman & Hall, London
Williamson M (2002) Alien plants in the British Isles. In: Pimentel D (ed) Biological invasions: economic and environmental costs of alien plant, animal, and microbe species. CRC Press, Boca Raton, pp 91–112
Williamson M, Pyšek P, Jarošík V, Prach K (2005) On the rates and patterns of spread of alien plants in the Czech Republic, Britain and Ireland. Ecoscience 12:424–433
Winter M, Schweiger O, Klotz S, Nentwig W, Andriopoulos P, Arianoutsou M, Basnou C, Delipetrou P, Didžiulis V, … Kühn I (2009) Plant extinctions and introductions lead to phylogenetic and taxonomic homogenization of the European flora. Proceedings of the National Academy of Sciences of the USA 106:21721–21725