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Is it worth the effort? Spread and management success of invasive alien plant species in a Central European National Park

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Abstract

The management of invasive alien species (IAS) in protected areas has become increasingly important in recent years. In this study, we analyse IAS management in the bilateral National Park Thayatal-Podyjí at the Austrian-Czech border. Based on two surveys from the years 2001 and 2010 and on annual management data from 2001-2010 we analyse changes in distribution and the efficiency of IAS management of three invasive alien plants (*Fallopia × bohemica*, *Impatiens glandulifera*, *Robinia pseudoacacia*).

In 2010, the three study species had invaded 161 ha (2%) of the study area. Despite a decade of management, *F. × bohemica* has become widespread, whereas *I. glandulifera* distribution has decreased strongly. The most widespread species, *R. pseudoacacia*, has declined substantially in cover, but the area invaded has increased.

From 2001 to 2010, annual management effort declined by about half. Management effort per hectare and decade was highest for *F. × bohemica* (2,657 hours), followed by *R. pseudoacacia* (1,473 hours) and *I. glandulifera* (270 hours). Management effort for achieving the same amount of reduction in population size and cover was highest for *R. pseudoacacia*, followed by *F. × bohemica* and *I. glandulifera*.

We conclude that substantial effort and resources are necessary to successfully manage the study species and have to be provided over prolonged time periods, and thus continued management of these species is recommended. We highly recommend a systematic approach for monitoring the efficiency of IAS management projects in protected areas.

Keywords

Conservation, eradication, floodplains, invasion, monitoring, nature conservation, protected areas

Introduction

The number of alien species are rapidly increasing worldwide, causing large and increasingly detrimental impacts on biodiversity and human well-being (Vilà et al. 2010, 2011). Protected areas play a pivotal role for nature conservation, and this is particularly the case in Europe as this continent is characterized by a long history of human impact on ecosystems. Recent studies have shown that protected areas are vulnerable to the spread of alien species (Foxcroft et al. 2013). Although only a small fraction of these become invasive, i.e. cause negative impacts on the environment (Blackburn et al. 2014) by outcompeting native species, changing ecosystem functioning and processes, or modifying species' interactions (Hulme et al. 2012, Pyšek et al. 2012), their impacts on the environment may be substantial (Foxcroft et al. 2013). Thus, the need for managing IAS in protected areas is high (Pyšek et al. 2013, Schmiedel et al. 2015, 2015, Sitzia et al. 2016). To date, however, analyses of IAS management effort and success in protected areas are scarce (see Foxcroft et al. 2013 and references therein), but urgently needed for informing evidence-based conservation.

In this study, we analyse management success of three invasive alien plant species (*Fallopia* × *bohemica*, *Impatiens glandulifera*, *Robinia pseudoacacia*) in the National Park Thayatal-Podyjí, over the period of a decade. These species had previously been identified by the management authority of being likely the most detrimental alien species in this protected area (C. Übl pers. comm.). We use spatially explicit distribution data from the years 2001 and 2010, and data of management effort and allocation from 2001 to 2010, to ask the following questions: (1) What is the change in distribution (extent and cover) of the study species? (2) What are the differences in trajectories of distribution change of managed and unmanaged populations, and between species? (3) How successful and efficient is the management? Finally, we discuss the implications of our study for IAS management in protected areas.

Material and methods

Study area

The National Park Thayatal-Podyjí is located in the Bohemian Massif on both sides of the Austrian-Czech Border (Fig. 1). It covers an area of 7,630 ha (Austria: 1,330 ha, Czech Republic: 6,300 ha) and it was established in 2000 (Austria) and 1991 (Czech Republic) to protect a heavily forested steep river valley. The Thaya River is the main water course in the National Park, whose narrow, meandering valley forms the border between Austria and the Czech Republic. The climate of the region is temperate, with cool winters and warm summers, average annual temperature ranging between c. 9.0°C in the lowest parts to 7.5°C in the highest parts, and annual precipitation of c. 500–600 mm (1961-90) (Chytry and Grulich 1993, ZAMG 2001). The bedrock consists of granite and gneiss which are partly covered by loess (Chytry et al. 1999, Nagl 2002).

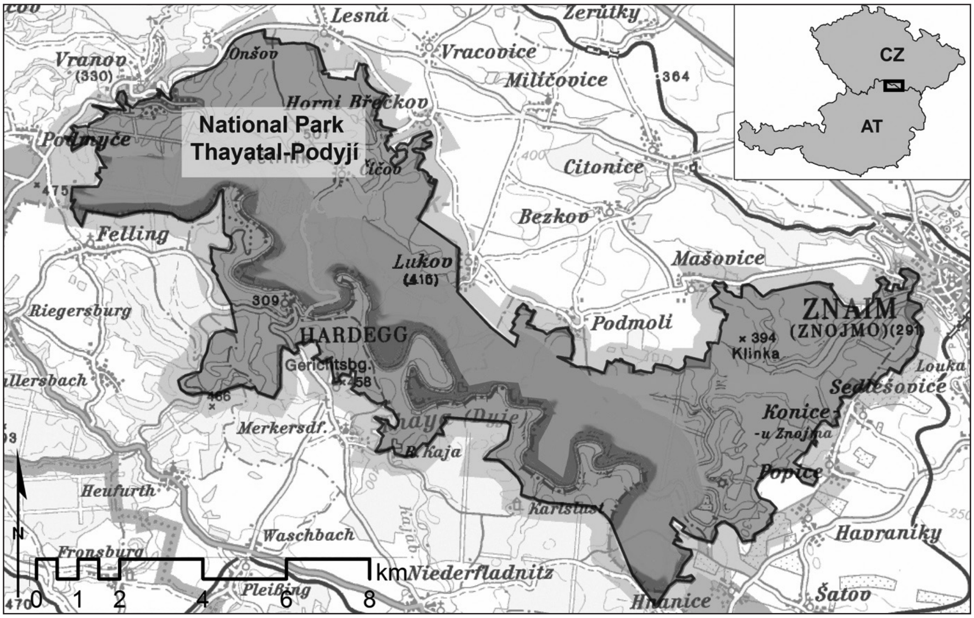

Figure 1. The National Park Thayatal-Podyjí (in grey) located at the Austrian-Czech border. The river Thaya/Dyje forms the border between the two countries. The insert shows the location of the National Park in Austria/Czech Republic.

Besides widespread zonal broadleaved-forests (mostly dominated by *Carpinus betulus*, *Fagus sylvatica*, *Quercus petraea*, *Q. robur*), rare and threatened habitats such as dry forests on steep slopes, floodplain-forests along the Thaya River, nutrient poor mesic and dry grasslands, contribute to the high value for nature conservation. After World War II, the Iron Curtain prevented land use intensification and ensured the conservation of this highly valuable landscape.

Study species

This study focuses on three study species which exert substantial negative impacts on native biota in Central Europe (DAISIE 2013).

Fallopia × *bohemica* (Polygonaceae) is a hybrid of *F. japonica* and *F. sachalinensis*, both native to East-Asia. The parental species were introduced into Europe in the 19th century for ornamental purposes, with the hybrid most probably originating in Europe (Bailey 1988). In the last decades, *F.* × *bohemica* spread vigorously (Mandák et al. 2004, Pauková 2013). It can establish dense populations, particularly in riparian habitats (Hejda et al. 2009, Pauková 2013), which change the vegetation structure and outcompete native species (Aguilera et al. 2010, Gerber et al. 2008, Stoll et al. 2012). Rhizome fragments of *Fallopia*-taxa are easily dispersed by running water during floods

(Beerling et al. 1994, Sheppard et al. 2006, Bímová et al. 2004) or contaminated soil. Due to the high regenerative ability of *Fallopia*-taxa (Bímová et al. 2004, Pauková 2013), management presents a substantial challenge (Rudenko and Hulting 2010, Delbart et al. 2012).

Impatiens glandulifera (Balsaminaceae) is native in the Himalayas. It was introduced as an ornamental plant to England in 1839. Despite its early introduction, only in the last few decades has it become one of the most widespread alien species in Europe (Beerling and Perrins 1993, Essl and Rabitsch 2006). It favours moist and nutrient-rich riparian habitats (Pyšek and Prach 1995). As an annual plant species it is reliant on seed dispersal, again often by water and contaminated soils (e.g. Dawson and Holland 1999). The impacts of *I. glandulifera* on native flora and fauna are contested (Drescher and Prots 1996, Tickner et al. 2001, Hejda and Pyšek 2006, Tanner et al. 2014). For instance, it has been shown that it may monopolize pollinators, thus likely lowering seed production of native plants (Chittka and Schürkens 2001), and that it exerts negative impacts on invertebrate communities (e.g. bugs, beetles) (Tanner et al. 2013). On the other hand, pollinators use its flowers as important nectar resource (Bartomeus et al. 2010).

Robinia pseudoacacia (Fabaceae) is a pioneer tree native to south-eastern North America. It has been extensively planted in Central Europe since the end of the 18th century, and has become widely naturalized in warm lowlands. Once established, *R. pseudoacacia* spreads efficiently via root suckers (e.g. Kowarik 2010, Cierjacks et al. 2013). As a consequence, the further local spread of *R. pseudoacacia* from invasion foci is difficult to control. *Robinia pseudoacacia* prefers nutrient poor dry and semi-dry habitats (e.g. dry grasslands and dry forests; Kleinbauer et al. 2010). Due to its symbiotic nitrogen fixing nodule bacteria, its encroachment into nutrient poor habitats severely increases productivity and modifies nutrient cycling (Rice et al. 2004, Kowarik 2010, Cierjacks et al. 2013).

Distribution mapping and data on management effort

The distribution of the study species within the Austrian part of the National Park Thayatal-Podyjí was first surveyed in 2000 (Essl and Hauser 2002). Distribution maps of the study species and aerial photos of the Natura 2000-site "Thayatal bei Hardegg", which includes the National Park Thayatal-Podyjí, were used for the survey. All populations of the study species were assigned to size classes (0–10 m^2; 10–100 m^2; 100–1,000 m^2; 1,000–10,000 m^2; >10,000 m^2) and population density was assessed according to the cover-abundance-scale of Braun-Blanquet (1964). For each population, the management options were assessed according to the local situation (i.e. population size, accessibility, the likelihood of further spread), and taking into account the nature conservation value of the invaded habitat; this assessment served as basis for subsequent management. In the second survey, all populations were resurveyed in summer 2010. Supplementary distribution data of the study species for the Czech part of the National Park were provided (R. Stejskal pers. comm.), which date from 2007–2013 (*F.* × *bohemica*, *I. glandulifera*), and 2003 (*R. pseudoacacia*).

We obtained data on management effort for the study species from the National Park Thayatal-Podyjí management authority for the years 2001, 2008, 2009 and 2010. Information provided included monthly working hours (differentiated into project-coordination and field-work) for each of the study species. Equipment and the amount of time of its usage were also recorded. For the period 2002–2007, data on management effort were incomplete. Thus, we used the available data from the National Park Authority, which showed that overall management effort changed gradually from year to year. We then interpolated data by assuming a linear change in management effort per study species from 2002–2007 based on data for the years for which complete information was available. We do acknowledge however, that this interpolation of management effort may introduce some uncertainties in the overall calculation of species specific management efforts.

Data analyses

Distribution data of the study species were entered into a database and a Geographical Information system (ESRI ArcView). For the Austrian part of the National Park, information on cover and size of the populations were available for both surveys. For each species, we analysed the changes in the spatial extent of polygons (Fig. 2) and in average cover values. For the latter, the cover-abundance data (according to Braun-Blanquet 1932, 1964) were converted into numerical cover values (Van der Maarel 1979) (r = 0.005%; + = 0.1%; 1 = 5%; 2 = 17.5%; 3 = 37.5%; 4 = 67.5%; 5 = 87.5%). We note that this approach is limited in its analytical power (as values are transformed into a standard numerical value for each Braun-Blanquet class), but nevertheless, it is the most appropriate method for analysing plot data (Van der Maarel 2007).

In addition, we calculated a combined index which gives equal weight to population size and cover. This "Area-Density-Index" ("ADI") was calculated by multiplying the extent of the polygons (in m^2) by the average plant cover (in %) for each population. We then calculated the ADI for each species by summing up all population-level ADIs; this was done separately for both surveys.

Management effort for the three study species between the two surveys (2001–2010) was calculated based on data from the National Park Authority. Again, these data were only available for the Austrian part of the National Park, and analyses thus excludes the Czech National Park-section. Management data provided include the number of working hours spent and the allocation of these hours to the study species per year. We calculated overall management effort for each study species by summing the annual working hours from 2001–2010. As management was done mostly manually and the necessary machinery was already available in the National Park Management Authority, we discard additional costs for machinery and equipment. Therefore, our calculation of management effort is conservative.

To analyse the efficiency of management, we used as a common metric the reduction of the ADI between both surveys. Only managed populations were considered,

Figure 2. Changes in distribution of *Fallopia* × *bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* between the first (2001) (**A**) and the second survey (2010) (**B**) in the surroundings of the village Hardegg. This section of the National Park Thayatal-Podyjí is the most heavily invaded.

whereas populations which were not managed were excluded. Finally, we calculated the number of working hours necessary for a reduction of the ADI by 10 points between both surveys; again, this was done separately for each study species.

Results

Current distribution of the study species

The study species differ strongly in their abundance and distribution in the study area (Suppl. material 3, 4). *Fallopia* × *bohemica* was by far the rarest species, with a total of 21 populations in 2010. Of these, eight are located in Austria (total area: 2,700 m^2) and the remainder (total area: 300 m^2) in the Czech part of the National Park. Thirteen populations invade ruderal habitats and tall herb vegetation near settlements, whereas eight populations are situated in riparian habitats (mostly in tall herb vegetation). Five populations are dense with cover values >50%. However, as the spatial extent of the populations is generally low, the ADI is also low.

In 2010, *I. glandulifera* was uncommon as well. In total, 90 populations with an extent of 1.2 ha were recorded whereby most of these (78) were located in the Czech part of the National Park. All populations were situated in near-natural habitats (tall herb vegetation, riparian forests) adjacent to the Thaya River. Populations in Austria are larger (up to 2,500 m^2), whereas the largest population in the Czech part covers only 300 m^2. Population cover values were mostly low (0.1–5%), and therefore the ADI is low.

In 2010, *R. pseudoacacia* was the most widespread study species. A total of 229 populations were recorded, of which 204 were situated in the Czech Republic. *Robinia pseudoacacia* invades 146 ha, whereby the Austrian part contributes only 13.8 ha. Most populations are found in dry oak-forests, a few populations have been recorded in dry grasslands. Population sizes vary substantially, with 41 populations being larger than one ha. Plant cover in the Austrian part of the National Park varied widely between low (5%) and high (three populations with >50%) values. For the Czech Republic, no information on plant cover was available. Managed populations were girdled, but due to re-sprouting, *R. pseudoacacia* still remains present in the herb and shrub layers.

Changes in abundance between 2001 and 2010

Numbers and extent of *F.* × *bohemica* populations increased between both surveys. Seven populations were found in 2001 and eight in 2010; four of these were newly established ones, while three populations were eradicated between both surveys (Suppl. material 1, 2). The ADI of the only population being present in both surveys decreased due to a major decline in plant cover, while the ADI of the other populations combined remained almost unchanged (Fig. 3). In total, *F.* × *bohemica* populations increased by c. 2,000 m^2 in size, and populations being present in both surveys increased by c. 300 m^2 or 54% (Suppl. material 3).

In contrast, numbers and extent of *I. glandulifera* population decreased (Suppl. material 1, 2). Ten populations were eradicated, whereas five new populations were found in 2010. Consequently, the ADI declined strongly as well (Fig. 3). This is mostly due to a large decrease in size, as the total area invaded by *I. glandulifera* (4.5 ha in 2001) decreased by 77% (-3.5 ha) in 2010. Densities of populations recorded in both surveys remained largely unchanged at a low level (Suppl. material 3, Suppl. material 1, 2).

Finally, *R. pseudoacacia* populations showed opposing trends. Whereas the number of populations and their extent increased, densities declined moderately. Of the 21 populations recorded in 2001, three were eradicated and six were newly recorded in 2010 (Suppl. material 3). The trend in ADI differs between managed and unmanaged populations with strongest declines for managed populations (Fig. 4).

Management measures, effort and efficiency

In total 6.4 ha (~0.1% of the National Park area) invaded by one of the three study species were managed between 2001 and 2010 (Suppl. material 3). Of these, 0.06 ha

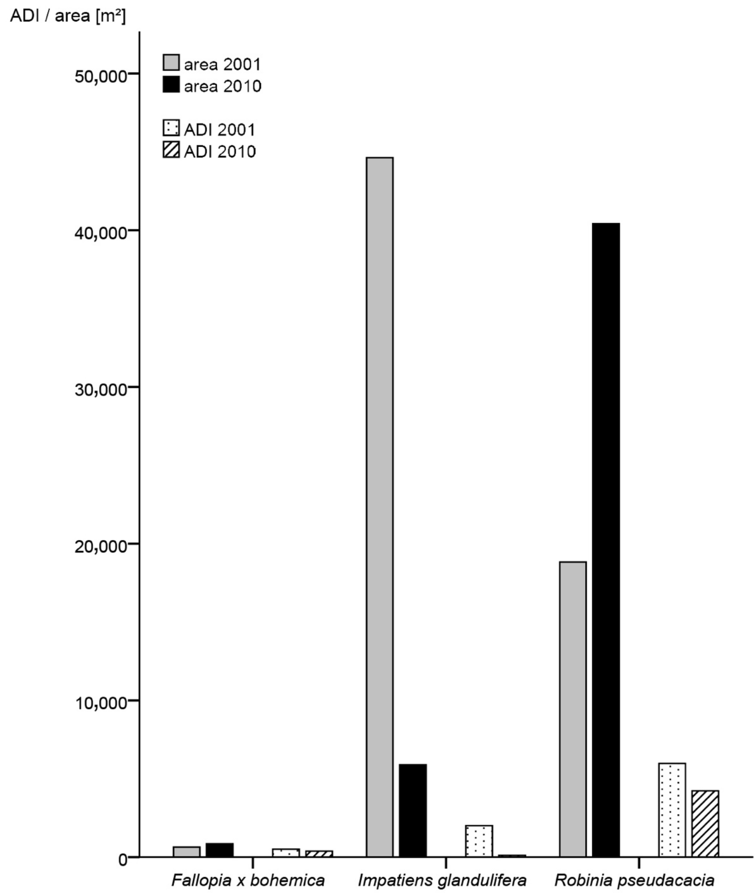

Figure 3. Total invaded area [m^2] and Area-Density-Index (ADI) of *Fallopia* × *bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the first (2001) and second survey (2010) in the Austrian part of the National Park Thayatal-Podyjí.

(1%) were invaded by *F.* × *bohemica*, 4.5 ha (77%) by *I. glandulifera* and 1.8 ha (28%) by *R. pseudoacacia*. Management of the study species varied in terms of methods and effort applied. *Fallopia* × *bohemica* populations were managed in varying ways and from different parties (road maintenance department, commune, National Park Management Authority). One population was not managed, one was mown once a year, and two were mown and had herbicides (Roundup, with the active ingredient Glyphosate) applied several times via stem-injections. Of the three eradicated populations, one was mown several times a year, the second was dug up, and one small population was removed manually by continuous hand-pulling. All populations of *I. glandulifera* were managed since 2001. Management measures include mowing once before the onset of flowering, small populations were managed by hand-pulling. *Robinia pseudoacacia* was managed by girdling at breast-height, leaving a small section

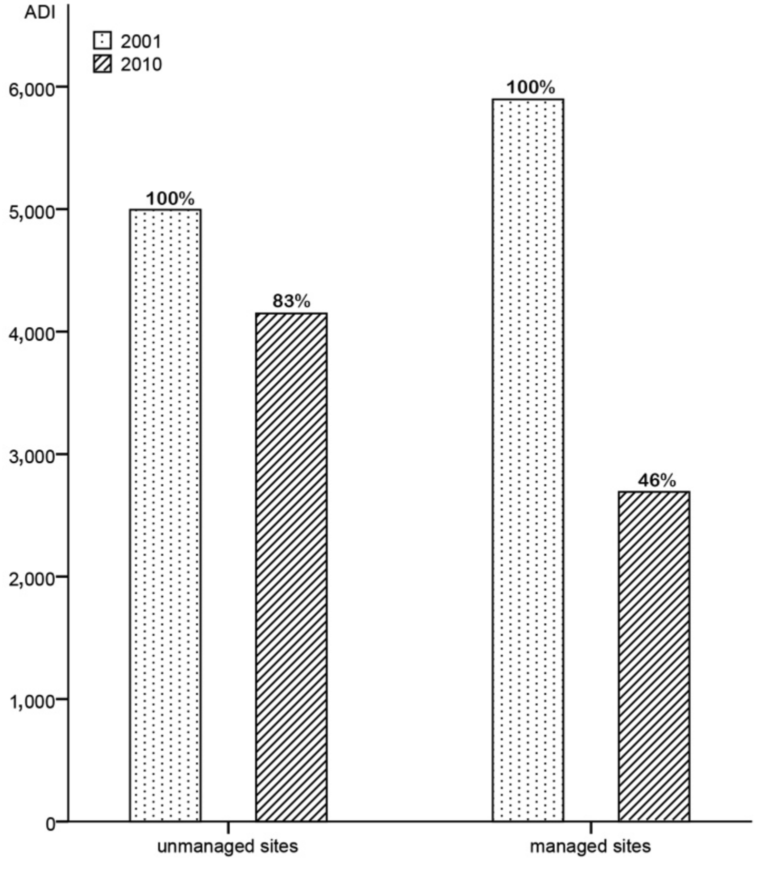

Figure 4. Change of the Area-Density-Index (ADI) of unmanaged and managed populations of *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí between the first (2001) and second survey (2010).

of the bark, which was removed in the following year. Root suckers were cut several times in subsequent years.

A total of 4,150 working hours were spent for the management of the study species during the ten-year period. On average, 16 hours were spent annually for managing *F.* × *bohemica*, 122 hours for *I. glandulifera* and 277 hours for *R. pseudoacacia*. Annual management effort declined by about half between the two surveys. Accordingly, in the first five-year period (2001–2005) 503 working hours were spent on average annually for managing the study species, whereas in the second five-year period (2006–2010) this value declined to 328 hours (Table 1). Management effort per hectare and decade was highest for *F.* × *bohemica* (2,657 hours), followed by *R. pseudoacacia* (1,473 hours) and *I. glandulifera* (270 hours). Management effort required for achieving the same reduction in plant cover and extent (calculated as reduction by 10 ADI-points), was highest for *R. pseudoacacia* (16 hours), followed by *F.* × *bohemica* (12 hours) and *I. glandulifera* (6 hours) (Table 2).

Table 1. Estimated annual working hours used for managing *Fallopia × bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí. Note that incomplete data on management effort for the years 2002-07 were interpolated (see main text for details).

	F. × bohemica hours	*I. glandulifera* hours	*R. pseudoacacia* hours	TOTAL
2001	21	163	372	**557**
2002	20	155	354	**530**
2003	19	147	336	**503**
2004	18	140	318	**476**
2005	17	132	300	**449**
2006	16	124	282	**422**
2007	15	116	264	**395**
2008	12	117	239	**368**
2009	11	70	170	**250**
2010	9	55	139	**203**
TOTAL	**159**	**1217**	**2774**	**4151**

Table 2. Estimated management effort and management efficiency for *Fallopia × bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí. Given are the total change in ADI (=Area-Density-Index, calculated by multiplying the extent of populations (in m^2) by the average plant cover (in %) for each species) of managed populations between the first (2001) and second survey (2010), the total number of working hours spent for management per study species, the total extent of managed populations at the start of management (in 2001), the working hours per decade and ha, and the numbers of working hours necessary to achieve a decrease of 10 ADI-points between both surveys. Note that incomplete data on management effort for the years 2002-07 were interpolated (see main text for details).

	F. × bohemica	*I. glandulifera*	*R. pseudoacacia*
ADI - change 2001/2010	-129	-1889	-1750
Working hours/10y	159	1217	2774
Area managed since 2001 (ha)	0.06	4.50	1.88
Hours/ha/10y	**2657**	**270**	**1473**
Hours/10 pts ADI-decrease/10y	**12**	**6**	**16**

Discussion

Changes of distribution and cover

Although the study species are amongst the most abundant IAS in Central Europe (Drescher and Prots 1996, Mandák et al. 2004, Chytry et al. 2008, Cierjacks et al. 2013), they are still rather uncommon in the study area. This moderate level of invasion can likely be attributed to a low level of anthropogenic disturbance, low regional human population density and low propagule pressure (Essl et al. 2012) due to the location of the study region along the former Iron Curtain.

The changes in population sizes and densities of the three study species showed opposing trends between both surveys. Most populations of *I. glandulifera* were eradicated by 2010 and the ones still existing had strongly declined in ADI. As an annual herb *I. glandulifera* is dependent on reproduction by seeds, thus management measures can result in rapid reduction in population size (e.g. Dawson and Holland 1999).

Fallopia × *bohemica* mainly regenerates vegetatively via rhizomes. As a result, management of *F.* × *bohemica* is difficult, and the species increased its distribution in the national park. Even though Bímová et al. (2004) report that *F.* × *bohemica* is mostly found in river bank vegetation in the Czech Republic, the majority of records in the study area have been made in anthropogenic habitats away from watercourses. However, populations growing adjacent to watercourses likely function as source for downstream spread, as has already occurred in two cases between both surveys. Several populations recorded in 2010 were located downstream from the populations found in 2001, suggesting that propagule dispersal by running water played a role (Dawson and Holland 1999).

Similar to *F.* × *bohemica*, *R. pseudoacacia* extended its distribution, although populations show a moderate decrease of ADI, because of strong declines in population density. After disturbance (i.e. logging) it may regenerate by root suckers, and may therefore even increase the size of the invaded area. Hence, complete eradication of managed populations was not feasible during the ten-year period. In the study area, *R. pseudoacacia* occurs in deciduous dry forests and semi-dry grasslands. Populations in forests changed little in size between both surveys, whereas populations in grasslands did. Terwei et al. (2013) and Motta et al. (2009) identified light availability and disturbances as crucial factors facilitating *R. pseudoacacia* regeneration. This is consistent with the slower spread of populations in forests (compared to open grasslands) in the National Park Thayatal-Podyjí.

The role of management on distribution changes

Management of *F.* × *bohemica* reduced population size, but only chemical management and mowing several times a year was effective. In contrast, populations mown once a year increased in size and cover remained high. Increases in population extent were even higher than in unmanaged populations, indicating that mowing once a year is not a suitable management measure (Catford et al. 2012, Delbart et al. 2012).

The management of *I. glandulifera* was highly successful as nearly all populations in the National Park Thayatal-Podyjí were eradicated within a decade. Only few populations in the eastern part of the study area persisted, possibly due to propagule pressure from populations further upstream the Thaya River. Such newly established populations were detected soon while still small (often consisting only of few individuals) and were rapidly included in the management. The existence and scale of impacts of *I. glandulifera* invasions are, however, discussed controversially (e.g. Prowse 2001, Hulme and Bremner 2005, Bartomeus et al. 2010), and some studies found no negative impacts on species richness and composition (e.g. Bartomeus et al. 2010, Hejda and Pyšek 2006). In addition, Hejda and Pyšek (2006) and Hulme and Bremner (2005) indicate that control measures may pave the way for subsequent invasions of other alien species. In this context, we found in the second survey that several locations formerly occupied by *I. glandulifera* were invaded by *Solidago gigantea*, which had been largely absent in the first survey (Essl and Hauser 2002). This species may cause severe effects on soil properties and species assemblages in Central European habitats (Güsewell et al. 2006, Kowarik 2010, Koutika et al. 2011).

Managed populations of *R. pseudoacacia* decreased more strongly in ADI than unmanaged ones, but complete eradication of managed populations has not been achieved to date, as sprouting individuals are still frequent.

Implications for management of alien species in protected areas

Managing *Fallopia* spp. is difficult and costly, and thus early response is crucial for management success. For small populations eradication is achievable, whereas for large populations halting further spread is often the only option. *Fallopia* × *bohemica* exhibits a greater tolerance to clipping than its parental species (Rouifed et al. 2011), so mechanical treatment is less effective than application of herbicides (Delbart et al. 2012). The non-selective herbicides Imazapyr and Glyphosate are the most effective and most commonly used (Rudenko and Hulting 2010). Herbicide application is also the least laborious management measure (Delbart et al. 2012). However, under the current legislation in Central European countries, mechanical control often represents the only option, especially in protected areas. To conclude, we suggest that stem injection of herbicides be allowed in protected areas. However, future options for chemical control might become more limited because of increasing concerns of Glyphosate application. Potentially, the release of biocontrol agents for *Fallopia* spp. in Central Europe, as has already been done in Great Britain (Shaw et al. 2011), might be added to the portfolio of future management strategies. However, this option should only be used after rigorous host-specificity testing to avoid unwanted side-effects.

We found that management of *I. glandulifera* in large protected areas is feasible when the species is relatively rare. To achieve complete eradication continued monitoring of suitable habitats along river stretches close to the national park boundaries is important, to avoid re-colonization from populations outside the National Park Thayatal-Podyjí (Malíková and Prach 2010, Kowarik 2010). The management procedure used, i.e. one early clipping before flowering (June/early July) and manually removing overlooked plants in August, proved to be appropriate. As *I. glandulifera* is widespread in Central Europe, regional eradication will need long-term monitoring (Hejda and Pyšek 2006).

We found that even using the most effective management measures (girdling), complete eradication of managed populations of *R. pseudoacacia* is difficult to achieve within a decade. Our findings therefore emphasize that managing *R. pseudoacacia* needs to be done over prolonged time periods (Cierjacks et al. 2013), although we found that management effort decreases sharply within a few years. Experience from the Czech part of the National Park Thayatal-Podyjí suggests that a combined approach of girdling and grazing with goats is another successful management option in habitats where grazing is possible (e.g. dry grasslands). Other suitable methods are girdling and planting native tree species to provide shade in the future, and the injection of herbicides into the stem. As all management measures are costly and time-consuming, *R. pseudoacacia* populations situated in forests may best be controlled by minimising anthropogenic disturbances (Terwei et al. 2013). This approach, which is compatible with conservation goals in protected areas, is consistent with observations in the Czech part of the National Park, where old-growth *R. pseudoacacia*-populations are starting to collapse. Similarly, the ADI of managed as well as of unmanaged populations in forests decreased between the two surveys (Fig. 4).

Implications for managing IAS in protected areas

In this study, we analysed the study species management by using an Area-Density-Index (ADI). This metric has the advantage that it allows comparison between management efficiency using a standardized measure which considers changes in population size and cover values. We found substantial differences in management effort that are necessary for the same reduction in ADI between species, with *R. pseudoacacia* requiring the most, and *I. glandulifera* the least effort. Although context-specificity (e.g. difficult accessibility of some *R. pseudoacacia* populations on steep slopes, which increases management effort per area) affects these results, some general conclusions can be drawn. Managing perennial plant species which spread vigorously vegetatively is particularly difficult and after a decade of management, only few populations are eradicated. Both *F.* × *bohemica* and *R. pseudoacacia* are known to be particularly difficult to manage (Delbart et al. 2012, Cierjacks et al. 2013, Schmiedel et al. 2016), but also are IAS with the highest environmental impacts in Europe (DAISIE 2013).

Conclusions

We found that substantial resources provided over prolonged time periods are needed for effectively managing invasive alien plant species in protected areas. The paucity of quantitative data on management effort is a severe constraint for assessing the efficiency of IAS management (Delbart et al. 2012). This is unfortunate, as currently we lack a profound understanding on the efficiency of alien plant species management, in particular in protected areas (Pyšek et al. 2013) and over the long-term (Sitzia et al. 2016). To improve monitoring of the efficiency of IAS management, we highly recommend a systematic approach for data collection on management effort in IAS management projects in protected areas.

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Supplementary material 1

Figure S1. Distribution of the three study species (*F.* × *bohemica*, *I. glandulifera*, *R. pseudoacacia*) in the National Park Thayatal-Podyjí in 2010

Authors: Verena Schiffleithner, Franz Essl

Data type: images

Explanation note: *Robinia pseudoacacia* predominantly invades forests near settlements, *Impatiens glandulifera* the Thaya river valley, and *Fallopia* × *bohemica* occurs mostly near settlements close to streams.

Supplementary material 2

Figure S2. Distribution of the three study species, *F.* × *bohemica*, *I. glandulifera*, and *R. pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí in 2001

Authors: Verena Schiffleithner, Franz Essl

Data type: images

Supplementary material 3

Table S1. Populations of *Fallopia* × *bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí

Authors: Verena Schiffleithner, Franz Essl

Data type: species data

Explanation note: Populations of *Fallopia* × *bohemica*, *Impatiens glandulifera* and *Robinia pseudoacacia* in the Austrian part of the National Park Thayatal-Podyjí, indicating population size, changes in population size between both surveys, and if management was applied.

Supplementary material 4

Table S2. Percentage and numbers of populations of the three study species

Authors: Verena Schiffleithner, Franz Essl

Data type: species data

Explanation note: Percentage and numbers of populations of the three study species in the five size classes (1 = 0–10m², 2 = 10–100m², 3 = 100–1,000m², 4 = 1,000–10,000m², 5 = >10,000m²) in the National Park Thayatal-Podyjí in 2010. Density classes (according to Braun-Blanquet 1964) are provided for *F.* × *bohemica* and *I. glandulifera*.