Received: 12 April 2025 | Accepted: 6 September 2025

DOI: 10.1111/1365-2664.70183

Journal of Applied Ecology

RESEARCH ARTICLE

Biocontrol by a native hemiparasite: Rapid suppression of noxious invasive plants in a field experiment

Kateřina Knotková[1] | Tamara Těšitelová[1] | Adam Knotek[1,2] | Irena Axmanová[1] | Kryštof Chytrý[1,3] | Helena Chytrá[1] | Zuzana Plesková[1] | Zdenka Preislerová[1] | Jakub Těšitel[1]

[1]Department of Botany and Zoology, Faculty of Science, Masaryk University, Brno, Czech Republic

[2]Department of Botany and Biodiversity Research, University of Vienna, Vienna, Austria

[3]Department of Botany, Charles University, Prague, Czech Republic

Correspondence
Kateřina Knotková
Email: knotkova@sci.muni.cz

Funding information
Grantová Agentura České Republiky, Grant/Award Number: 21-22488S

Handling Editor: Romina Fernandez

Abstract

1. Alien plant invasions are a recognised threat to biodiversity. Although experimental evidence remains limited, their success may be reduced by parasitic plants, which have been proposed as promising biocontrol agents.
2. We tested the effects of hemiparasitic *Melampyrum arvense* on *Soligago gigantea* and *Symphyotrichum lanceolatum*, clonal herbaceous species from North America, which are noxious invaders of European vegetation. In a multi-site, 4-year field experiment, we compared the effects of the hemiparasite presence combined with mowing and mowing alone to the unmown controls.
3. The experimental treatments significantly affected invasive hosts' cover, height and below-ground biomass. Mowing alone significantly reduced host growth; however, hemiparasites had a stronger effect, reducing median host cover from 90% and 78% to 8% and 12% after 3 years in *Solidago* and *Symphyotrichum*, respectively. Growth reduction was faster in *Solidago* than in *Symphyotrichum* under the same treatments. Hemiparasites also severely reduced the host's below-ground biomass by more than 80% compared to the unmown treatment.
4. Mowing increased community diversity and species turnover, with significantly stronger effects when combined with hemiparasites. However, the increase in native grassland species was limited, likely due to a restricted local species pool.
5. *Synthesis and applications*. *Melampyrum arvense* was demonstrated as a biocontrol tool suitable for containing two harmful invasive species. After the suppression of the invaders, an additional introduction of native species by seeding or green-hay transfer could facilitate community restoration.

KEYWORDS
biological control, biotic resistance, hemiparasites, invasive plants, *Melampyrum arvense*, mowing, *Solidago gigantea*, *Symphyotrichum lanceolatum*

This is an open access article under the terms of the Creative Commons Attribution License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.
© 2025 The Author(s). *Journal of Applied Ecology* published by John Wiley & Sons Ltd on behalf of British Ecological Society.

1 | INTRODUCTION

Biodiversity loss is a key aspect of global change, threatening natural ecosystems and disrupting ecosystem services (Bacher et al., 2023). Invasive alien plant species are one of the major drivers of biodiversity loss, displacing native species and disrupting ecosystem functions (Pyšek et al., 2004). They alter habitats by outcompeting natives, modifying nutrient cycling, increasing productivity, and causing social and economic damage, including resource insecurity, health risks, reduced land value and loss of recreational and aesthetic values (Bacher et al., 2023; Pyšek & Richardson, 2010; Vilà et al., 2011). Various strategies can be used to manage invasive species (Pergl et al., 2016). Complete eradication is ideal but typically feasible only in early stages or isolated sites with low reinvasion risk. Otherwise, it may require intensive effort and yield only temporary results. Tolerance may be necessary when invaders are widespread or resources are limited, even for harmful species. A commonly recommended compromise is a stratified approach—combining containment in priority natural areas with tolerance in less valuable sites, such as urban or synanthropic habitats. This allows for targeted management and improved cost-efficiency, though continuous effort is required due to reinvasion risk.

Among available control methods are mechanical approaches (e.g. topsoil removal), chemical control (herbicide application) and suppression through biocontrol. The latter builds on ecological theories to exploit natural enemies such as predators, parasites and pathogens of invasive species. The success of invasive species is often linked to the absence of such enemies in the invaded area, as described by the Enemy Release Hypothesis (Maron & Vilà, 2001). This hypothesis underlies traditional biocontrol, which introduces specialist enemies from the invader's native range but carries risks of unintended consequences, including secondary invasions (de Clercq et al., 2011). In contrast, the Biotic Resistance Hypothesis (Levine & D'Antonio, 1999) proposes that invasive species may be constrained by antagonistic interactions with native generalist enemies, and specialist enemies are usually absent. Such interactions, mainly via herbivores and competitors, have been shown to reduce invasive plant abundance by limiting their establishment and vigour (Levine et al., 2004). Biocontrol based on the Biotic Resistance Hypothesis avoids introducing non-native species, can target expansive native species (Axmanová et al., 2024; Prach & Wade, 1992), and may create novel biotic interactions with strong impacts due to disequilibrium dynamics and limited potential for resistance (Cogni, 2010).

Parasitic plants have recently been recognised as potential biotic resistance agents in several studies across the globe (Těšitel et al., 2021). In Australia, *Cassytha pubescens* suppresses invasive shrubs such as *Ulex europaeus*, reducing biomass by up to 60%, while native species remain largely unaffected (Cirocco et al., 2017). Similarly, *Cuscuta australis* reduced invasive plant growth by up to 40% in China and promoted native biodiversity (Yu et al., 2011). Central European hemiparasites have also shown low host specificity and the ability to form functional haustoria on various invasive species, indicating biotic resistance potential (Knotková et al., 2024). The most comprehensive evidence for their biocontrol ability comes from multi-site experiments showing hemiparasitic *Rhinanthus* spp. suppress the expansive grass *Calamagrostis epigejos*, restore vegetation diversity and structure, and trigger turnover in arthropod communities in Europe (Těšitel et al., 2017, 2025). However, similar complex evidence for hemiparasites' biocontrol capacity on alien invasives, based on robust field experiments, vegetation monitoring and comparisons to standard restoration management, such as mowing, is missing.

To address the knowledge gap in hemiparasite biocontrol capacity, we established a multi-site field experiment testing the effect of the hemiparasite *Melampyrum arvense* (Orobanchaceae) on plant communities invaded by the alien clonal perennials *Solidago gigantea* and *Symphyotrichum lanceolatum* (Asteraceae). We selected *Melampyrum arvense* based on previous studies demonstrating its ability to attach to the roots of both invaders (Knotková et al., 2024) and reduce their above-ground biomass in a pot experiment (Těšitelová et al., 2024). Over 4 years, we monitored vegetation composition, species cover and invader height in three treatments: mowing, mowing combined with hemiparasite presence and unmown control plots. In the final year, we also assessed the viable below-ground biomass of the invaders. Previous studies paid no attention to the level of damage to the below-ground storage organs, which are crucial for plant survival. We hypothesized that the hemiparasite would have a stronger suppressive effect on the invaders' above-ground and below-ground biomass than mowing alone, with a particularly pronounced reduction in below-ground biomass. Additionally, we expected that hemiparasites' presence would alter plant community composition by increasing species diversity and facilitating the recovery and dominance of native species.

2 | MATERIALS AND METHODS

2.1 | Study species

Solidago gigantea and *Symphyotrichum lanceolatum* are tall perennials native to North America (Pyšek et al., 2012). They were introduced to Europe in the 19th century (Jedlička & Prach, 2006; Weber & Jakobs, 2005). Both species are considered transformer invaders, causing significant ecosystem change and biodiversity decline, and are classified as invasive species with a massive environmental impact (category BL2; Cubino et al., 2022; Hejda et al., 2021; Pergl et al., 2016; Richardson et al., 2000). They are spread by wind-dispersed seeds and underground rhizomes (Jedlička & Prach, 2006; Weber & Jakobs, 2005). Rhizomes grow in autumn and produce shoots each spring, with *Solidago* flowering in late summer and *Symphyotrichum* in autumn, while all above-ground biomass dies back in autumn (Schmid & Bazzaz, 1987). They typically colonize moist, nutrient-rich habitats such as wetlands, brownfields, old fields and disturbed meadows in Europe (Pyšek et al., 2012; Weber & Jakobs, 2005). Both species can be suppressed by regular ploughing or herbicides, though these

methods harm native species and are unsuitable for conservation sites (Szymura et al., 2022). Long-term control through mowing or grazing—optionally combined with topsoil removal, flooding and seeding of native species—reduces *Solidago* dominance and enhances biodiversity (Nagy et al., 2020; Szymura et al., 2022). For *Symphyotrichum lanceolatum*, empirical experience with conservation management is deficient.

Melampyrum arvense is an annual generalist hemiparasite attaching to host root systems (Matthies, 2017). It is native to Europe (Pyšek et al., 2022), with a distribution from France to the Urals and from the Mediterranean to Southern Sweden and Finland. It is considered threatened in the northwestern part of its range (Grulich, 2017; Leimu, 2010). It predominantly grows on calcareous substrates in steppes and dry meadows, along roadsides, in scrublands, and on field margins of warm regions (Těšitel et al., 2015).

2.2 | Experimental design

Experiments were established at six localities dominated by *Solidago gigantea* or *Symphyotrichum lanceolatum* in the southeastern Czech Republic in mid-June 2020 and continued to 2023. Soil pH at the study sites ranged from moderately acidic to slightly alkaline, with values between 6.2 and 7.5 (Data S1). At each site, we established four to eight experimental blocks. Each block consisted of three plots with the following treatments: Unmown (control), Mown (mowing with biomass removal twice a year—in late July and October), and Mown + *Melampyrum* (mowing twice a year plus *Melampyrum* sown in the first autumn). Altogether, we established 66 plots for *Solidago* and 54 plots for *Symphyotrichum*. The plots were 1.5 m × 1.5 m big (except for 1 m × 1 m plots at one site), separated by 1 m gaps. The treatments were applied on a broader area (0.25 m in each direction) to avoid edge effects. We thank the landowners and managers for access to the study sites: Lesy České republiky, the municipality of Kostice, the Czech Society for Ornithology, the Pálava PLA, and the Podyjí NP. We also thank the Podyjí NP Administration for research permission (permit no. NPP 1513/2019).

Melampyrum seeds were sourced from Liščí kopec near Sedlec, Czech Republic (GPS: 48°45′47.6″N, 16°42″32.6″ E), and Havranické vřesoviště in the National Park Podyjí near Havraníky, Czech Republic (48°48′52.7″N, 16°0″6.3″ E). Whole plants with ripening seeds were collected in July 2020 (and 2021) and dried. Seeds released from capsules were collected and stored at room temperature. 300 seeds per 1 m^2 were sown onto the Mown + *Melampyrum* treatment plots. Sowing was done in October 2020. Plots with less than 30% *Melampyrum* cover in June 2021 were reseeded in October 2021 (Data S1). Before sowing, plots were raked to remove litter and ensure seed-soil contact. Mown plots received the same raking treatment as the Mown + *Melampyrum* plots.

We monitored the percentual cover of the target invasive species (further called 'hosts'), host height (mean height of host plants in a plot), and vegetation composition recorded with the Braun-Blanquet cover-abundance scale (Dengler et al., 2008). Species richness (number of species) and Simpson diversity index were computed from the vegetation data as univariate diversity measures. The monitoring started with baseline data collection before the onset of treatments and continued yearly in three consecutive seasons. As a result, these data come from a replicated before-after-control-impact design, which is considered one of the strongest experimental designs allowing causal inference (Christie et al., 2019).

In mid-July 2023, in the final year of the experiment, we collected below-ground biomass of rhizomes and roots. Five samples were taken from each plot using soil cores (depth 10 cm; diameter 5 cm) at fixed positions arranged in a quincunx pattern. Living rhizomes and roots of the invasive species were extracted from the samples, washed, dried at 60°C and weighed. The resulting values of below-ground host biomass were subsequently scaled to 1 m^2 by dividing the mass by the number of cores (5) and their area size ($\pi \times 0.025^2$).

2.3 | Statistical analysis

Analyses were conducted separately for each invasive species using identical analytical approaches. We used linear mixed-effect models to analyse experimental treatments' effects on the hosts' cover and height and community diversity indices. The models included treatment, time and their interaction as fixed effect terms. The interaction term describing the differential temporal trends between treatments was indicative of treatment effect significance based on the repeated-measure structure of the data. We specified custom contrasts in the treatment factor to test the significance of regression coefficients for the comparisons Mown versus Unmown and Mown + *Melampyrum* versus Mown. The random effect term of the models included plot identity nested in the block, which was further nested in the locality. Species richness was square-rooted to improve the homogeneity of variance and normality of the residuals. Other variables were not transformed because visual inspections of the residuals indicated satisfactory alignment with the linear model assumptions (normality of the residuals and homogeneity of variance). We also tried models with other data transformations (sqrt and log) in these models, but none of them matched the assumptions better than the current models. The below-ground biomass data were only collected at the end of the experiment. Therefore, we used a simplified model structure containing only the treatment as a single fixed effect predictor and block identity nested in the locality as a random effect. Below-ground biomass data were square-rooted before analysis. Post hoc comparisons between the treatment levels were based on a t-test of the respective regression coefficients (significance) and on computing Cohen's *d* standardised difference (effect size), which was computed for temporal trends in the case of interactive effects or for main effects on host underground biomass.

Community composition was analysed using distance-based redundancy analyses (RDAs) based on square-rooted Bray–Curtis dissimilarity. Species cover data were log $(x+1)$ transformed before analysis. The effects of the experimental treatments were tested by a Monte–Carlo permutation test with a split-plot design and 999

permutations. Their significance was determined as the significance of conditional effects of dummy-coded treatment × time terms in the RDAs, where blocks at individual localities and treatment and time main effects were included as covariates. See Data S3 for full details of the RDA analyses.

All analyses were conducted in R (version 4.4.2; R Core Team, 2024). Mixed-effect and multivariate models were fitted in packages nlme (version 3.1; Pinheiro et al., 2024) and vegan (Oksanen et al., 2024), respectively. Effect sizes were computed in package emmeans (Lenth, 2024). All data used in the analyses are publicly available from Zenodo (Knotková et al., 2025; https://doi.org/10.5281/zenodo.16541333).

3 | RESULTS

3.1 | Effect of mowing and *Melampyrum* on the hosts

Melampyrum was established in all sown plots (Mown + *Melampyrum* treatment), with cover ranging from 2% to 50% in the first year. Over time, *Melampyrum* cover mainly increased, and seeds naturally dispersed into the surrounding areas. In the third year of the experiment, it also started expanding in the Mown treatment (Data S2).

Over the 4-year experimental period, the cover and height of *Solidago* remained stable in the Unmown control (Figure 1a,b). Mowing significantly reduced both cover and height compared to Unmown control plots (Table 1; Figure 1a,b). Further significant reductions were observed under the Mown + *Melampyrum* treatment (Table 1; Figure 1a,b). In *Symphyotrichum*, the observed effects were similar but slightly less pronounced than in *Solidago* (Figure 1c,d).

The experimental treatments had a significant effect on the below-ground biomass of both *Solidago* and *Symphyotrichum* (Table 1; Figure 2). The below-ground biomass values of both hosts were significantly lower under the Mown treatment compared to the Unmown control, and further lower under the Mown + *Melampyrum* treatment.

3.2 | Effect of mowing and *Melampyrum* on vegetation diversity and composition

The tested treatments significantly affected the species richness and Simpson index over time (Table 1; Figure 3). The species

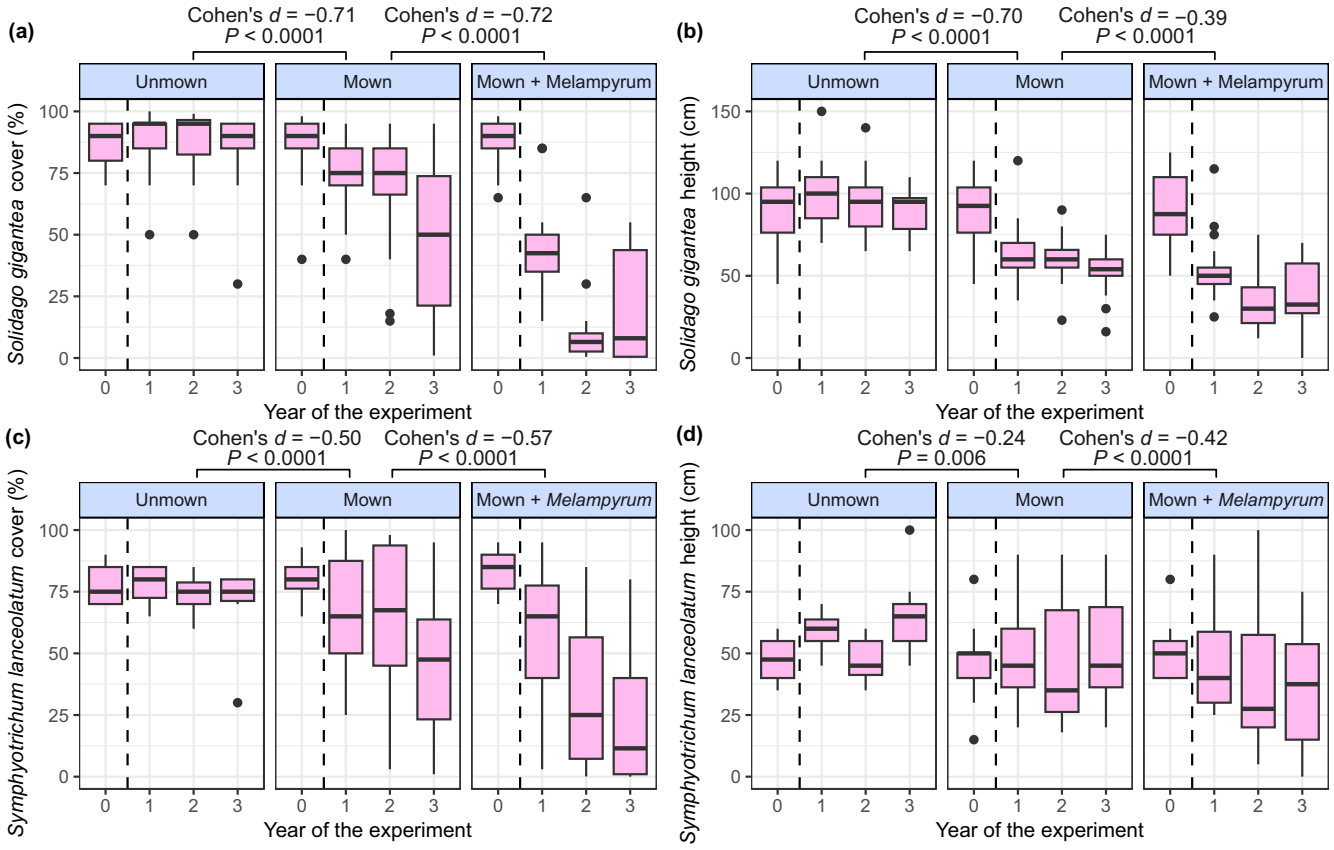


FIGURE 1 Effects of experimental treatments on the cover and height of *Solidago gigantea* (a, b) and *Symphyotrichum lanceolatum* (c, d) over 4 years. Boxplots display medians, quartiles, and non-outlier ranges, with outliers as points. Vertical dashed lines indicate the start of treatments. Cohen's *d* values indicate standardised differences between the temporal trends of the treatments. *p*-values indicate the significance of the differences based on contrast *t*-tests. In the last year of the experiment, *Melampyrum* naturally spread into the Mown treatment (Data S2).

Response	Predictor	df	*F*-value	*p*-value
Solidago gigantea				
Cover	Treatment	2,42	159.59	<0.0001
	Time	1,183	169.68	<0.0001
	Treatment×time	2,183	51.10	<0.0001
Height	Treatment	2,42	134.23	<0.0001
	Time	1,183	100.14	<0.0001
	Treatment×time	2,183	30.62	<0.0001
Underground biomass	Treatment	2,34	51.53	<0.0001
Species richness	Treatment	2,42	19.25	<0.0001
	Time	1,183	213.83	<0.0001
	Treatment×time	2,183	39.45	<0.0001
Simpson index	Treatment	2,42	39.70	<0.0001
	Time	1,183	161.35	<0.0001
	Treatment×time	2,183	33.04	<0.0001
Symphyotrichum lanceolatum				
Cover	Treatment	2,26	34.36	<0.0001
	Time	1,135	129.62	<0.0001
	Treatment×time	2,135	19.37	<0.0001
Height	Treatment	2,26	32.64	<0.0001
	Time	1,135	0.64	0.4247
	Treatment×time	2,135	7.89	0.0006
Underground biomass	Treatment	2,87	9.10	0.0019
Species richness	Treatment	2,26	1.73	0.1968
	Time	1,135	67.16	<0.0001
	Treatment×time	2,135	12.54	<0.0001
Simpson index	Treatment	2,26	5.87	0.0079
	Time	1,135	128.53	<0.0001
	Treatment×time	2,135	8.44	0.0004

TABLE 1 Analysis of variance tables summarize the effects of experimental treatments on invaders, including cover, height, below-ground biomass, diversity and Simpson index.

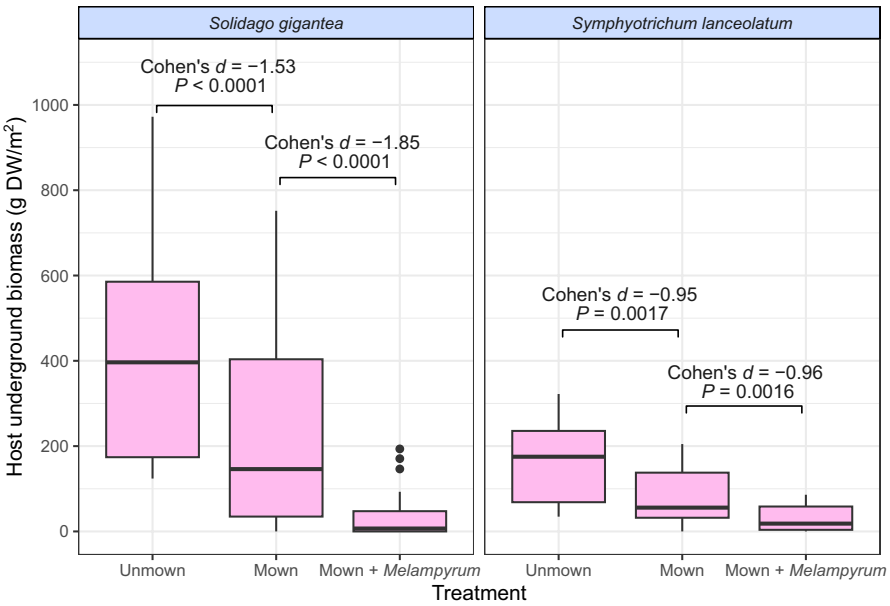


FIGURE 2 Effects of experimental treatments on the below-ground biomass of invasive species (rhizomes and roots), collected in the final year of the experiment. Cohen's *d* values indicate standardized differences between the treatments. *p*-values indicate the significance of the differences based on contrast *t*-tests. Boxplots display medians, quartiles and non-outlier ranges, with outliers as points.

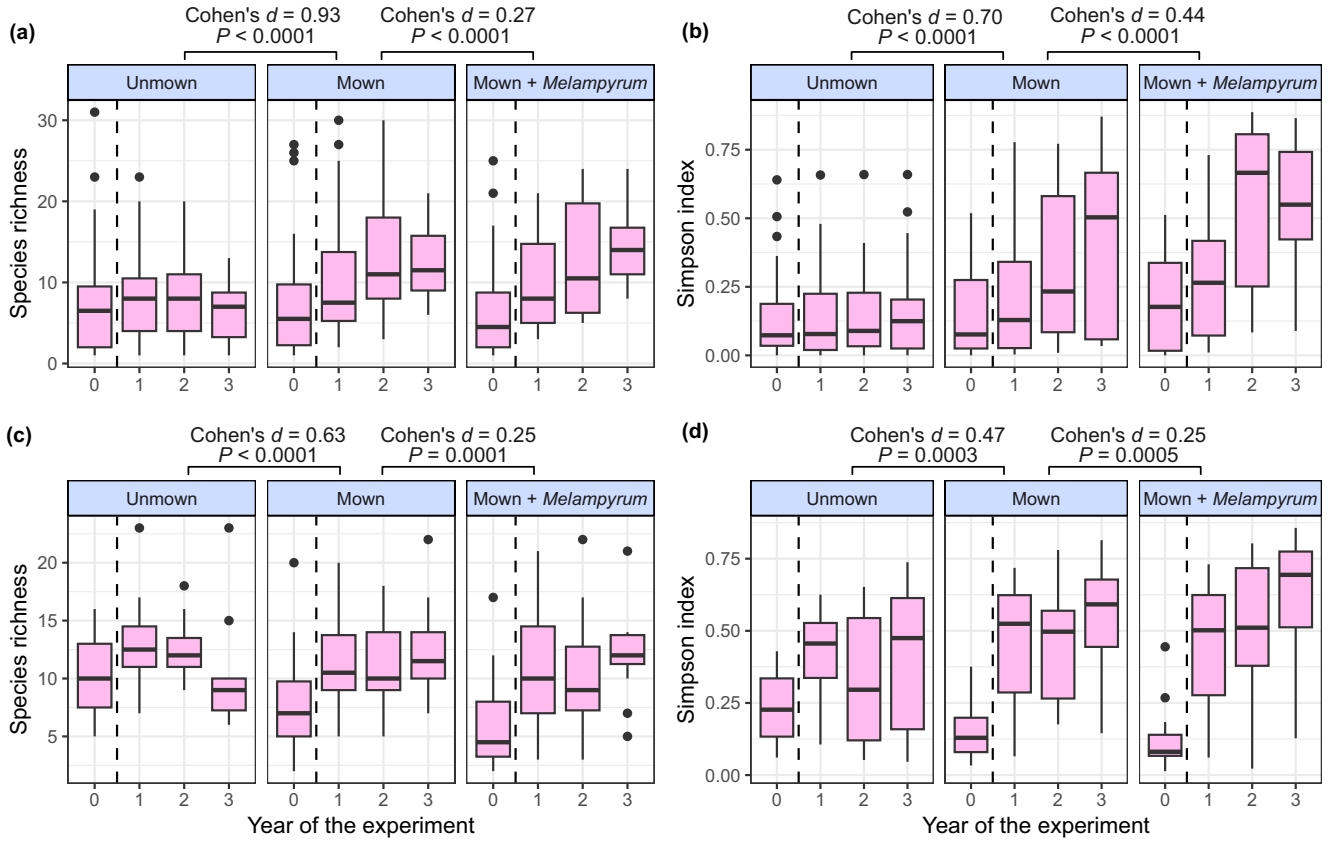


FIGURE 3 Effects of experimental treatments on *Solidago gigantea* (a, b) and *Symphyotrichum lanceolatum* (c, d) richness (a, b) and Simpson index (c, d) over 3 years. Cohen's *d* values indicate standardized differences between the temporal trends of the treatments. *p*-values indicate the significance of the differences based on contrast *t*-tests. Boxplots display medians, quartiles, and non-outlier ranges, with outliers as points. Vertical dashed lines indicate the start of treatments.

richness remained stable for *Solidago* and *Symphyotrichum* in the Unmown control. However, the species richness increased significantly for both *Solidago* and *Symphyotrichum* in the Mown treatment compared to Unmown controls, and further increased under the Mown + *Melampyrum* treatment (Figure 3a,c). Simpson's diversity index displayed similar trends. Its low values were stable in *Solidago* Unmown control plots, while a slightly increasing trend could be seen in *Symphyotrichum* controls (Figure 3b,d). In the Mown treatment, the Simpson index increased significantly in both *Solidago* and *Symphyotrichum* compared to Mown plots, and further increased under the Mown + *Melampyrum* treatment (Figure 3b,d).

Monte–Carlo permutation tests (999 permutations) of conditional effects in distance-based RDAs revealed significant directional effects of the experimental treatments on the community composition in both *Solidago*- (Mown × time: pseudo-$F = 2.35$, $p = 0.001$, Mown + *Melampyrum* × time: pseudo-$F = 7.02$, $p = 0.001$) and *Symphyotrichum*-dominated stands (pseudo-$F = 2.13$, $p = 0.017$, Mown + *Melampyrum* × time: pseudo-$F = 3.86$, $p = 0.001$). This directional trend was largely based on the decline of the invasive species and the relative increase of various common generalist species (Data S3).

4 | DISCUSSION

4.1 | Suppressive effects of hemiparasites and mowing on the invaders

Results of our experiment demonstrate the strong suppressive effects of hemiparasitic *Melampyrum arvense* and mowing on both invasive species. The combined effect of the two measures was drastic on *Solidago gigantea* and moderately strong on *Symphyotrichum lanceolatum* abundance. In both cases, the suppressive effect of the hemiparasite was stronger than the mowing management alone (mowing twice per season), even though in the last year, *Melampyrum* escaped from the sown plots and partly colonised also the mown-only plots at some of the research sites, likely due to myrmecochory (Figure 4). Ant-induced seed spread is difficult to avoid under natural conditions. This deviation in the experimental treatments likely affected host growth, reducing the contrast between Mown and Mown + *Melampyrum* treatments, potentially leading to an underestimation of the hemiparasite effect. However, treatment differences were significant in all parameters (Table 1).

Solidago is known to be relatively responsive to various management practices. Szymura et al. (2022) demonstrated a gradual

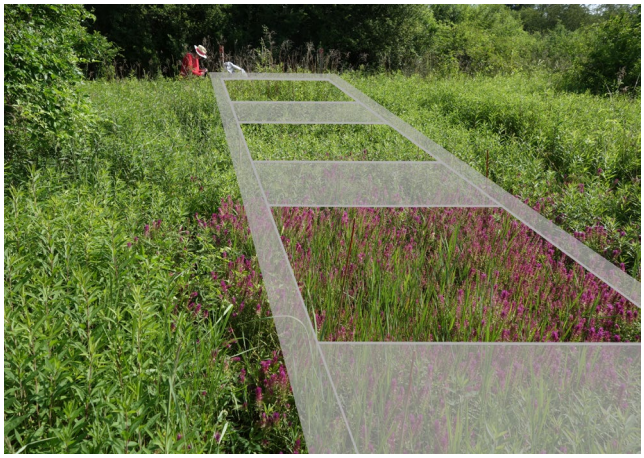


FIGURE 4 Illustrative example of a block with treatments: Unmown (background), Mown, and Mown + *Melampyrum* (foreground). Buffer zones in transparent grey. Image from 2022 (Year 2).

decline in its cover from nearly 100% to 25% over 6 years under regular mowing applied twice per season. A weak decline observed after 2 years of mowing in the same study is comparable to our results (*approx*. 20% reduction in mown plots after 2 years). However, mowing did not completely eradicate *Solidago*, even when combined with an intense initial disturbance (Nagy et al., 2020; Szymura et al., 2022). While similar or stronger suppression of *Solidago* may be achieved through long-term cattle or sheep grazing or flooding, these approaches require extended time and specific site conditions (Nagy et al., 2020). By contrast, *Melampyrum* reduced *Solidago* cover in established vegetation by nearly 90% within 2 years (Figure 1). Comparable suppression was observed in a pot experiment, where substantial biomass reduction occurred after 1 year of cultivation (Těšitelová et al., 2024). Although *Melampyrum* is a generalist hemiparasite parasitizing various forbs (Knotková et al., 2024; Matthies, 2017), the fitness it achieves depends on the host species, as host quality varies among species (Matthies, 2017). In a pot experiment (Těšitelová et al., 2024), *Solidago* was classified as a highly suitable host compared to the native species tested by Matthies (2017), which is also supported by thriving *Melampyrum* in *Solidago*-dominated stands in our experiment (Data S2).

Symphyotrichum was classified as a moderately suitable host for *Melampyrum*, as reflected by lower parasite biomass and a reduction in host biomass by only one-third in a pot experiment (Těšitelová et al., 2024). Correspondingly, the suppressive effect of the Mown + *Melampyrum* treatment was more gradual than that observed for *Solidago*, resulting in a 70% cover decline after 3 years (Figure 1). Nevertheless, the effect of *Melampyrum* remained substantially stronger than mowing alone, which led to only a 43% reduction (Figure 1). Despite the widespread invasion of *Symphyotrichum* across Europe (Axmanová et al., 2021) and its documented negative impacts on vegetation (Hejda et al., 2021; Lanta et al., 2022; Pergl et al., 2016), field studies evaluating management options have focused exclusively on drastic measures such as herbicide application or regular tillage (Chmielewski & Semple, 2001).

The effect of the treatments was even more pronounced below-ground. Root and rhizome biomass of both invaders was reduced by nearly 50% by mowing alone (Figure 2). A similar effect of mowing was reported for invasive *Solidago altissima* (Stoll et al., 1998) and was attributed to the limited production of new rhizomes (Meyer & Schmid, 1999a). Clonal plants are generally adapted to disturbances of intermediate severity and frequency by storing carbohydrates in below-ground organs, which enables resprouting (Klimešová et al., 2021; Martínková et al., 2020). Regular disturbance (e.g. mowing or grazing) can gradually deplete these storage reserves (Bartušková et al., 2022), while hemiparasites impose additional resource stress. In our study, this combined pressure reduced below-ground biomass by 90% in *Solidago* and 80% in *Symphyotrichum* relative to the Unmown plots. Clonal species are disproportionally more affected by parasitism, likely due to resource sharing through the rhizome network, which facilitates efficient hemiparasite exploitation (Demey et al., 2015; Mudrák & Lepš, 2010). Our study is the first to directly demonstrate the drastic effects on host rhizome biomass, correlating with the strong decline in above-ground cover. Similar levels of above-ground suppression by hemiparasites have been reported for *Calamagrostis epigejos* (Těšitel et al., 2025) and *Carex acuta* (Decleer et al., 2013), but without corresponding data on below-ground biomass. Our results show that after suppressing the above-ground growth, the studied invasive species do not persist as below-ground rhizomes. This elimination of the storage and resprouting organs is crucial for the long-term suppression of clonal invaders.

4.2 | Hemiparasite's effects on diversity and native species abundance

Melampyrum presence increased community diversity and promoted the abundance of numerous grassland species (e.g. *Plantago lanceolata*, *Hypericum perforatum*, *Picris hieracioides*; Data S3). However, many of the supported species are not considered restoration targets. For instance, *Melampyrum* promoted some competitive grasses, consistent with its limited ability to parasitise grasses (Matthies, 2017). Various ruderal and early-successional species (e.g. *Papaver rhoeas*, *Descurainia sophia*, *Carduus acanthoides*), including the invasive *Erigeron annuus*, increased in *Melampyrum* and/or mown plots (Data S3), likely colonising open space after the targeted invaders' decline. This suggests that additional measures may be needed to support the restoration of target communities. These measures may include seeding target species or green-hay transfer to overcome the insufficiency of the seed bank and dispersal limitation. The same additional-measure requirement was used in a previous study, where mowing, herbicide, or mechanical disturbance was tested as a control measure for *Solidago* (Szymura et al., 2022). This contrasts with spontaneous community recovery and the increase of target species observed after the

suppression of *Calamagrostis epigejos* by *Rhinanthus* spp. (Galvánek et al., 2024; Těšitel et al., 2025). In contrast to our experimental sites, *Calamagrostis* frequently spreads to high-conservation-value grasslands, where endangered species may persist for some time, even under its moderate *Calamagrostis* dominance (Somodi et al., 2018). While *Solidago* and *Symphyotrichum* can also invade such grasslands, they more commonly colonise disturbed sites of lower conservation value (Cubino et al., 2022; Hejda et al., 2021; Richardson et al., 2000). That was also the case at most of the research sites included in our study. These findings highlight the importance of ecological context when predicting community development following restoration and considering the need for additional measures, such as seeding or hay transfer.

4.3 | *Melampyrum* application, risks, and limitations

The suppression of both invaders by *Melampyrum* was strong and, for *Solidago*, even more effective than the effect of *Rhinanthus* on the expansive grass *Calamagrostis epigejos* (Těšitel et al., 2025). Notably, *Rhinanthus* has already been applied in ecological restoration in the Czech Republic (Lukavský, 2020). *Melampyrum* was efficient not only on dry sites matching its ecological preference but also on mesic, moist, and even occasionally flooded habitats. This broad habitat tolerance resembles *Rhinanthus alectorolophus*, which thrived on *Calamagrostis epigejos* even outside its optimal conditions. However, it declined following its host's decline (Galvánek et al., 2024; Těšitel et al., 2017).

The application of *Melampyrum arvense* in ecological restoration by sowing is straightforward. Seeds must be sown in autumn to break dormancy through cold stratification (Matthies, 2017; Oesau, 1975). This procedure is similar to *Rhinanthus* applications; however, *Melampyrum* may germinate in autumn and overwinter as a germinated seed (epicotyl dormancy; Oesau, 1975). Its larger seed size and earlier germination may reduce the sensitivity of *Melampyrum* to spring drought, a known limiting factor for *Rhinanthus* establishment (Ameloot et al., 2006; Mudrák et al., 2014). Lower seed density may be sufficient (compare 300 seeds m^{-2} used here to 500 seeds m^{-2} recommended for *Rhinanthus*; Mudrák et al., 2014). However, seed production in *Melampyrum* is challenging. Seeds are released continuously over an extended period, making harvest less efficient than in *Rhinanthus* (Hejduk S., personal communication 2025). Despite limited seed availability, *Melampyrum* remains a promising restoration tool thanks to its efficient short-distance myrmecochory (Winkler & Heinken, 2007). It can gradually spread across a site under suitable management, even when sown in small patches, as indicated by colonisation of mown-only plots in our experiment (Data S2). Late mowing (late July or August) may support *Melampyrum* seed production while suppressing the reproduction of *Solidago* and *Symphyotrichum* by cutting just before flowering (Meyer & Schmid, 1999b).

The use of *Melampyrum arvense* is generally safe from a human health perspective. Although it was formerly a weed in winter cereal fields, this is no longer an issue in modern agriculture (Gislén, 1949). Moreover, long-distance dispersal from the restoration sites is very unlikely (Winkler & Heinken, 2007). *Melampyrum arvense* was also demonstrated to be positively associated with high vegetation diversity and generally missing from low-diversity vegetation (Fibich et al., 2017). However, concerns about potential negative biodiversity effects have been raised as Heer et al. (2021) identified a negative correlation between high *Melampyrum arvense* cover (>ca 30%) and plant species richness. Nevertheless, the densities of *Melampyrum arvense* observed in the current study (Data S2) exceeded this threshold only in the second year after sowing. Its density dropped below 30% the following year in approximately three-quarters of the plots. This indicates a density-dependent relationship, likely driven by reduced host resources following the suppression of the invasive species (Data S2). This retreat appears to be slower than that observed for *Rhinanthus alectorolophus* after the suppression of *Calamagrostis*, where collapses of the hemiparasite population were observed (Galvánek et al., 2024; Těšitel et al., 2017). Long-term observation of sites restored using *Melampyrum* is currently lacking; however, this species may persist at restoration sites at lower densities, resulting in sustained suppression of invasive species and preventing their resurgence.

5 | CONCLUSIONS AND APPLICATIONS

Our research demonstrated hemiparasitic *Melampyrum arvense* as an efficient biotic resistance agent for two harmful plant invaders through suppressing their growth and both above- and below-ground biomass. The impact of *Melampyrum* was comparable with the most efficient current management practices for *Solidago gigantea* (grazing, herbicides). For *Symphyotrichum lanceolatum*, *Melampyrum* presents the first empirical evidence-based management option. The observed reduction of host below-ground biomass undermines the invasive clonal host strategy and suggests that the biocontrol effect could persist in the long term. The use of *Melampyrum* must be combined with mowing, which alone has only a limited impact on the suppression of the invaders. The introduction of target species by seeding or green-hay transfer of regional origin seems necessary for the efficient restoration of biodiversity in strongly impoverished, invaded communities. The ability of *Melampyrum* to spread spontaneously via myrmecochory across the restoration site may promote the gradual restoration of larger areas and support its persistence, enabling efficient local containment of invaders. Our research provides the most comprehensive evidence of the effects of biotic resistance in the suppression of selected plant invaders, promising great potential for practical application.

AUTHOR CONTRIBUTIONS

Kateřina Knotková and Jakub Těšitel conceived the ideas, designed the methodology, and analysed the data; Kateřina Knotková, Tamara Těšitelová, and Jakub Těšitel drafted the manuscript;

Kateřina Knotková and Adam Knotek managed the field experiment. Kateřina Knotková, Tamara Těšitelová, Adam Knotek, Irena Axmanová, Kryštof Chytrý, Helena Chytrá, Zuzana Plesková, Zdenka Preislerová, and Jakub Těšitel collected the data in the field, contributed critically to the drafts, and gave final approval for publication.

ACKNOWLEDGEMENTS

We thank Petr Hubatka, Terezie Chamráthová, Valérie Hanušová and Štěpánka Knotková for their assistance in the field. The research was supported by the Czech Science Foundation (project 21-22488S). Open access publishing facilitated by Masarykova univerzita, as part of the Wiley - CzechELib agreement.

CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest related to this work.

DATA AVAILABILITY STATEMENT

The data and R script supporting the results of this study are available in Zenodo at: 10.5281/zenodo.16541333 (Knotková et al., 2025).

ORCID

Kateřina Knotková https://orcid.org/0000-0001-6668-8977
Tamara Těšitelová https://orcid.org/0000-0001-6901-6359
Kryštof Chytrý https://orcid.org/0000-0003-4113-6564
Jakub Těšitel https://orcid.org/0000-0003-3793-3704

REFERENCES

Ameloot, E., Verheyen, K., Bakker, J. P., De Vries, Y., & Hermy, M. (2006). Long-term dynamics of the hemiparasite *Rhinanthus angustifolius* and its relationship with vegetation structure. *Journal of Vegetation Science*, *17*, 637–646.

Axmanová, I., Chytrý, K., Boublík, K., Chytrý, M., Dřevojan, P., Ekrtová, E., Fajmon, K., Hájková, P., Härtel, H., Hejda, M., Horáková, V., Jongepier, J. W., Kalníková, V., Kaplan, Z., Koutecký, P., Lustyk, P., Pergl, J., Prach, K., Pyšek, P., … Těšitel, J. (2024). Catalogue of expansive plants of the Czech Republic. *Preslia*, *96*, 299–327. https://doi.org/10.23855/preslia.2024.299

Axmanová, I., Kalusová, V., Danihelka, J., Dengler, J., Pergl, J., Pyšek, P., Večeřa, M., Attorre, F., Biurrun, I., Boch, S., Conradi, T., Gavilán, R. G., Borja, J.-A., Knollová, I., Kuzemko, A., Lenoir, J., Leostrin, A., Medvecká, J., Moeslund, J. E., … Chytrý, M. (2021). Neophyte invasions in European grasslands. *Journal of Vegetation Science*, *32*, e12994. https://doi.org/10.1111/jvs.12994

Bacher, S., Galil, B. S., Nuñez, M. A., Ansong, M., Cassey, P., Dehnen-Schmutz, K., Fayvush, G., Hiremath, A. J., Ikegami, M., Martinou, A. F., McDermott, S. M., Preda, C., Vilà, M., Weyl, O. L. F., Fernandez, R. D., & Ryan-Colton, E. (2023). Chapter 4: Impacts of invasive alien species on nature, nature's contributions to people, and good quality of life. In H. E. Roy, A. Pauchard, P. Stoett, & T. Renard Truong (Eds.), *Thematic assessment report on invasive alien species and their control of the intergovernmental science-policy platform on biodiversity and ecosystem services*. IPBES secretariat.

Bartušková, A., Lubbe, F. C., Qian, J., Herben, T., & Klimešová, J. (2022). The effect of moisture, nutrients and disturbance on storage organ size and persistence in temperate herbs. *Functional Ecology*, *36*, 314–325. https://doi.org/10.1111/1365-2435.13997

Chmielewski, J. G., & Semple, J. C. (2001). The biology of Canadian weeds. 113. *Symphyotrichum lanceolatum* (Willd.) Nesom [*Aster lanceolatus* Willd.] and *S. lateriflorum* (L.) Löve & Löve [*Aster lateriflorus* (L.) Britt.]. *Canadian Journal of Plant Science*, *81*, 829–849.

Christie, A. P., Amano, T., Martin, P. A., Shackelford, G. E., Simmons, B. I., & Sutherland, W. J. (2019). Simple study designs in ecology produce inaccurate estimates of biodiversity responses. *Journal of Applied Ecology*, *56*, 2742–2754. https://doi.org/10.1111/1365-2664.13499

Cirocco, R. M., Facelli, J. M., & Watling, J. R. (2017). Does nitrogen affect the interaction between a native hemiparasite and its native or introduced leguminous hosts? *New Phytologist*, *213*, 812–821. https://doi.org/10.1111/nph.14181

Cogni, R. (2010). Resistance to plant invasion? A native specialist herbivore shows preference for and higher fitness on an introduced host. *Biotropica*, *42*, 188–193. https://doi.org/10.1111/j.1744-7429.2009.00570.x

Cubino, J. P., Těšitel, J., Fibich, P., Lepš, J., & Chytrý, M. (2022). Alien plants tend to occur in species-poor communities. *NeoBiota*, *73*, 39–56. https://doi.org/10.3897/neobiota.73.79696

de Clercq, P., Mason, P. G., & Babendreier, D. (2011). Benefits and risks of exotic biological control agents. *BioControl*, *56*, 681–698. https://doi.org/10.1007/s10526-011-9372-8

Decleer, K., Bonte, D., & Van Diggelen, R. (2013). The hemiparasite *Pedicularis palustris*: "Ecosystem engineer" for fen-meadow restoration. *Journal for Nature Conservation*, *21*, 65–71. https://doi.org/10.1016/j.jnc.2012.10.004

Demey, A., De Frenne, P., Baeten, L., Verstraeten, G., Hermy, M., Boeckx, P., & Verheyen, K. (2015). The effects of hemiparasitic plant removal on community structure and seedling establishment in semi-natural grasslands. *Journal of Vegetation Science*, *26*, 409–420. https://doi.org/10.1111/jvs.12262

Dengler, J., Chytrý, M., & Ewald, J. (2008). Phytosociology. In S. E. Jørgensen & B. D. Fath (Eds.), *Encyclopedia of ecology* (Vol. 4. General Ecology, pp. 2767–2779). Elsevier.

Fibich, P., Lepš, J., Chytrý, M., & Těšitel, J. (2017). Root hemiparasitic plants are associated with high diversity in temperate grasslands. *Journal of Vegetation Science*, *28*, 184–191. https://doi.org/10.1111/jvs.12472

Galvánek, D., Uhliarová, E., Ujházy, K., Ujházyová, M., Janišová, M., & Těšitel, J. (2024). Reversing expansion of *Calamagrostis epigejos* in Carpathian mountain grasslands by mowing and application of hemiparasitic *Rhinanthus alectorolophus*. *Applied Vegetation Science*, *27*, 1–12. https://doi.org/10.1111/avsc.12782

Gislén, T. (1949). Problems concerning the occurrence of *Melampyrum arvense* in Sweden. *Oikos*, *1*, 208–234. https://doi.org/10.2307/3564713

Grulich, V. (2017). The red list of vascular plants of the Czech Republic. *Příroda*, *35*, 75–132.

Heer, N., Klimmek, F., Kurtogullari, Y., Prati, D., Rieder, N. S., & Boch, S. (2021). Density effects of two hemiparasitic *Melampyrum* species on grassland plant diversity. *Tuexenia*, *41*, 411–422. https://doi.org/10.14471/2021.41.014

Hejda, M., Sádlo, J., Kutlvašr, J., Petřík, P., Vítková, M., Vojík, M., Pyšek, P., & Pergl, J. (2021). Impact of invasive and native dominants on species richness and diversity of plant communities. *Preslia*, *93*, 181–201. https://doi.org/10.23855/PRESLIA.2021.181

Jedlička, J., & Prach, K. (2006). A comparison of two North-American asters invading in central Europe. *Flora*, *201*, 652–657. https://doi.org/10.1016/j.flora.2006.01.002

Klimešová, J., Ottaviani, G., Charles-Dominique, T., Campetella, G., Canullo, R., Chelli, S., Janovský, Z., Lubbe, F. C., Martínková, J., & Herben, T. (2021). Incorporating clonality into the plant ecology research agenda. *Trends in Plant Science*, *26*, 1236–1247. https://doi.org/10.1016/j.tplants.2021.07.019

Knotková, K., Cempírková, H., & Těšitel, J. (2024). Native root hemiparasites form haustorial attachments with multiple invasive and expansive species. *Flora*, *319*, 152584. https://doi.org/10.1016/j.flora.2024.152584

Knotková, K., Těšitelová, T., Knotek, A., Axmanová, I., Chytrý, K., Chytrá, H., Plesková, Z., Preislerová, Z., & Těšitel, J. (2025). *Data from: Biocontrol by a native hemiparasite: rapid suppression of noxious invasive plants in a field experiment*. Zenodo https://doi.org/10.5281/zenodo.16541333

Lanta, V., Liancourt, P., Altman, J., Černý, T., Dvorský, M., Fibich, P., Götzenberger, L., Hornych, O., Miklín, J., Petřík, P., Pyšek, P., Čížek, L., & Doležal, J. (2022). Determinants of invasion by single versus multiple plant species in temperate lowland forests. *Biological Invasions*, *24*, 2513–2528. https://doi.org/10.1007/s10530-022-02793-8

Leimu, R. (2010). Habitat quality and population size as determinants of performance of two endangered hemiparasites. *Annales Botanici Fennici*, *47*, 1–13. https://doi.org/10.5735/085.047.0101

Lenth, R. (2024). "*emmeans: Estimated Marginal Means, aka Least-Squares Means* (R package version 1.10.5)".

Levine, J. M., Adler, P. B., & Yelenik, S. G. (2004). A meta-analysis of biotic resistance to exotic plant invasions. *Ecology Letters*, *7*, 975–989. https://doi.org/10.1111/j.1461-0248.2004.00657.x

Levine, J. M., & D'Antonio, C. M. (1999). Elton revisited: A review of evidence linking diversity and invasibility. *Oikos*, *87*, 15–26.

Lukavský, J. (2020). Funguje nám kokrhel? Aneb zkušenosti z regionu. *Ochrana Přírody*, 11–15.

Maron, J. L., & Vilà, M. (2001). When do herbivores affect plant invasion? Evidence for the natural enemies and biotic resistance hypotheses. *Oikos*, *95*, 361–373. https://doi.org/10.1034/j.1600-0706.2001.950301.x

Martínková, J., Klimeš, A., Puy, J., & Klimešová, J. (2020). Response of clonal versus non-clonal herbs to disturbance: Different strategies revealed. *Perspectives in Plant Ecology, Evolution and Systematics*, *44*(February), 125529. https://doi.org/10.1016/j.ppees.2020.125529

Matthies, D. (2017). Interactions between a root hemiparasite and 27 different hosts: Growth, biomass allocation and plant architecture. *Perspectives in Plant Ecology, Evolution and Systematics*, *24*, 118–137. https://doi.org/10.1016/j.ppees.2016.12.006

Meyer, A. H., & Schmid, B. (1999a). Experimental demography of rhizome populations of establishing clones of *Solidago altissima*. *Journal of Ecology*, *87*, 42–54. https://doi.org/10.1046/j.1365-2745.1999.00317.x

Meyer, A. H., & Schmid, B. (1999b). Seed dynamics and seedling establishment in the invading perennial *Solidago altissima* under different experimental treatments. *Journal of Ecology*, *87*, 28–41. https://doi.org/10.1046/j.1365-2745.1999.00316.x

Mudrák, O., & Lepš, J. (2010). Interactions of the hemiparasitic species *Rhinanthus minor* with its host plant community at two nutrient levels. *Folia Geobotanica*, *45*, 407–424. https://doi.org/10.1007/s12224-010-9078-1

Mudrák, O., Mládek, J., Blažek, P., Lepš, J., Doležal, J., Nekvapilová, E., & Těšitel, J. (2014). Establishment of hemiparasitic *Rhinanthus* spp. in grassland restoration: Lessons learned from sowing experiments. *Applied Vegetation Science*, *17*, 274–287. https://doi.org/10.1111/avsc.12073

Nagy, D. U., Rauschert, E. S. J., Henn, T., Cianfaglione, K., Stranczinger, S., & Pal, R. W. (2020). The more we do, the less we gain? Balancing effort and efficacy in managing the *Solidago gigantea* invasion. *Weed Research*, *60*, 232–240. https://doi.org/10.1111/wre.12417

Oesau, A. (1975). Untersuchungen zur Keimung und Entwicklung des Wurzelsystems in der *Gattung Melampyrum* L. (Scrophulariaceae). *Beiträge Zur Biologie Der Pflanzen*, *51*, 121–147.

Oksanen, J., Simpson, G., Blanchet, F., Kindt, R., Legendre, P., Minchin, P., O'Hara, R., Solymos, P., Stevens, M., Szoecs, E., Wagner, H., Barbour, M., Bedward, M., Bolker, B., Borcard, D., Carvalho, G., Chirico, M., De Caceres, M., Durand, S., & Weedon, J. (2024). "*vegan: Community Ecology Package* (R package version 2.6-8)".

Pergl, J., Sádlo, J., Petrusek, A., Laštuvka, Z., Musil, J., Perglová, I., Šanda, R., Šefrová, H., Šíma, J., Vohralík, V., & Pyšek, P. (2016). Black, Grey and Watch Lists of alien species in the Czech Republic based on environmental impacts and management strategy. *NeoBiota*, *28*, 1–37. https://doi.org/10.3897/neobiota.28.4824

Pinheiro, J., Bates, D., & R Core Team. (2024). "*nlme: Linear and Nonlinear Mixed Effects Models* (R package version 3.1-166)".

Prach, K., & Wade, P. M. (1992). Population characteristics of expansive perennial herbs. *Preslia*, *64*, 45–51.

Pyšek, P., Chytrý, M., Pergl, J., Sádlo, J., & Wild, J. (2012). Plant invasions in the Czech Republic: Current state, introduction dynamics, invasive species and invaded habitats. *Preslia*, *84*, 575–629.

Pyšek, P., & Richardson, D. M. (2010). Invasive species, environmental change and management, and health. *Annual Review of Environment and Resources*, *35*, 25–55. https://doi.org/10.1146/annurev-environ-033009-095548

Pyšek, P., Richardson, D. M., Rejmánek, M., Webster, G. L., Williamson, M., & Kirschner, J. (2004). Alien plants in checklists and floras: Towards better communication between taxonomists and ecologists. *Taxon*, *53*, 131–143. https://doi.org/10.2307/4135498

Pyšek, P., Sádlo, J., Chrtek, J., Chytrý, M., Kaplan, Z., Pergl, J., Pokorná, A., Axmanová, I., Čuda, J., Doležal, J., Dřevojan, P., Hejda, M., Kočár, P., Kortz, A., Lososová, Z., Lustyk, P., Skálová, H., Štajerová, K., Večeřa, M., … Danihelka, J. (2022). Catalogue of alien plants of the Czech Republic (3rd edition): Species richness, status, distributions, habitats, regional invasion levels, introduction pathways and impacts. *Preslia*, *94*, 447–577. https://doi.org/10.23855/PRESLIA.2022.447

R Core Team. (2024). *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing.

Richardson, D. M., Pyšek, P., Rejmánek, M., Barbour, M. G., Dane Panetta, F., & West, C. J. (2000). Naturalization and invasion of alien plants: Concepts and definitions. *Diversity and Distributions*, *6*, 93–107. https://doi.org/10.1046/j.1472-4642.2000.00083.x

Schmid, B., & Bazzaz, F. A. (1987). Clonal integration and population structure in perennials: Effects of severing rhizome connections. *Ecology*, *68*, 2016–2022.

Somodi, I., Vadkerti, Á., & Těšitel, J. (2018). Thesium linophyllon parasitizes and suppresses expansive *Calamagrostis epigejos*. *Plant Biology*, *20*, 759–764. https://doi.org/10.1111/plb.12723

Stoll, P., Egli, P., & Schmid, B. (1998). Plant foraging and rhizome growth patterns of *Solidago altissima* in response to mowing and fertilizer application. *Journal of Ecology*, *86*, 341–354. https://doi.org/10.1046/j.1365-2745.1998.00263.x

Szymura, M., Świerszcz, S., & Szymura, T. H. (2022). Restoration of ecologically valuable grassland on sites degraded by invasive *Solidago*: Lessons from a 6-year experiment. *Land Degradation & Development*, *33*, 1985–1998. https://doi.org/10.1002/ldr.4278

Těšitel, J., Chytrý, K., Vašíček, M., Blažek, P., Galvánek, D., Fajmon, K., Holá, E., Horník, J., Jiráská, Š., Machač, O., Mládek, J., Mudrák, O., Řehounková, K., Těšitelová, T., Turisová, I., & Malenovský, I. (2025). Hemiparasitic plants facilitate ecological restoration of encroached European grasslands. *Journal of Environmental Management*, *374*, 124120. https://doi.org/10.1016/j.jenvman.2025.124120

Těšitel, J., Fibich, P., De Bello, F., Chytrý, M., & Lepš, J. (2015). Habitats and ecological niches of root-hemiparasitic plants: An assessment based on a large database of vegetation plots. *Preslia*, *87*, 87–108.

Těšitel, J., Li, A. R., Knotková, K., McLellan, R., Bandaranayake, P. C. G., & Watson, D. M. (2021). The bright side of parasitic plants: What are they good for? *Plant Physiology*, *185*, 1309–1324. https://doi.org/10.1093/plphys/kiaa069

Těšitel, J., Mládek, J., Horník, J., Těšitelová, T., Adamec, V., & Tichý, L. (2017). Suppressing competitive dominants and community restoration with native parasitic plants using the hemiparasitic *Rhinanthus alectorolophus* and the dominant grass *Calamagrostis*

epigejos. *Journal of Applied Ecology*, *54*, 1487–1495. https://doi.org/10.1111/1365-2664.12889

Těšitelová, T., Knotková, K., Knotek, A., Cempírková, H., & Těšitel, J. (2024). Root hemiparasites suppress invasive alien clonal plants: Evidence from a cultivation experiment. *NeoBiota*, *90*, 97–121. https://doi.org/10.3897/neobiota.90.113069

Vilà, M., Espinar, J. L., Hejda, M., Hulme, P. E., Jarošík, V., Maron, J. L., Pergl, J., Schaffner, U., Sun, Y., & Pyšek, P. (2011). Ecological impacts of invasive alien plants: A meta-analysis of their effects on species, communities and ecosystems. *Ecology Letters*, *14*, 702–708. https://doi.org/10.1111/j.1461-0248.2011.01628.x

Weber, E., & Jakobs, G. (2005). Biological flora of central Europe: *Solidago gigantea* Aiton. *Flora*, *200*, 109–118. https://doi.org/10.1016/j.flora.2004.09.001

Winkler, E., & Heinken, T. (2007). Spread of an ant-dispersed annual herb: An individual-based simulation study on population development of *Melampyrum pratense* L. *Ecological Modelling*, *203*, 424–438. https://doi.org/10.1016/j.ecolmodel.2006.12.006

Yu, H., Liu, J., He, W. M., Miao, S. L., & Dong, M. (2011). *Cuscuta australis* restrains three exotic invasive plants and benefits native species. *Biological Invasions*, *13*, 747–756. https://doi.org/10.1007/s10530-010-9865-x

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Data S1. Characteristics of experimental localities and evidence of reseeded plots.

Data S2. *Melampyrum arvense* establishment.

Data S3. Community ordination analysis.

How to cite this article: Knotková, K., Těšitelová, T., Knotek, A., Axmanová, I., Chytrý, K., Chytrá, H., Plesková, Z., Preislerová, Z., & Těšitel, J. (2025). Biocontrol by a native hemiparasite: Rapid suppression of noxious invasive plants in a field experiment. *Journal of Applied Ecology*, *62*, 3514–3524. https://doi.org/10.1111/1365-2664.70183