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The invasion of Goldenrod (*Solidago canadensis*) does not affect the ant communities (Hymenoptera, Formicidae) in central Romania meadows

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Abstract

Solidago canadensis (Canadian goldenrod) is a widespread invasive plant species that has been shown to cause strong effects on abundance, diversity and species composition of various ground-dwelling invertebrates. The present study aimed to investigate the effects of invasion by *S. canadensis* on the ant communities in meadows with different exposure (southern and northern) in centre Romania. Ant sampling was carried out on the Târnavelor plateau in six paired sites (invaded and non-invaded areas, total N = 12) using Barber pitfall traps. Our results showed that ant richness and abundance were not affected by the presence of *S. canadensis*, while ant abundance was marginally higher in sites with northern exposure (p=0.06). Moreover, the composition of ant communities was also affected by the exposure of the sites, whilst the effect of invasion itself was not significant. In summary, our study suggests rather a very limited effect of the spread of the Canadian goldenrod on the ant communities in Central Romania meadows, while local microclimates may play a higher role. This contrasts with other studies that showed negative effects of this plant species on the ant and other Hymenoptera communities in other regions.

Keywords

Ant assemblages, composition, diversity, invasive species, Transylvania

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Introduction

The proliferation of invasive alien species poses a significant threat to global biodiversity and ecosystem function, with human-mediated dispersal accelerating their establishment beyond native ranges (Early et al. 2016; Gentili et al. 2021; Seebens et al. 2021). Amongst this invasive species, *Solidago canadensis* L. (Asteraceae), the Canadian goldenrod is a perennial species which poses a major threat to semi-natural habitats across Central Europe (Moroń et al. 2009; Fenesi et al. 2015; Świerszcz et al. 2024). It has a long history on the continent, dating back to its introduction to England in 1645 where it was first planted as an ornamental plant (Poljuha et al. 2024). Today, the species is widespread and thrives in disturbed areas such as forest edges, roadsides, and meadows (Fenesi et al. 2015; Gallardo et al. 2019; Poljuha et al. 2024).

As expected, *Solidago canadensis* is affecting the native plant species (Fenesi et al. 2015). The species produces dense stands, which can reach over two meters, creating a dense canopy that reduces light penetration and dramatically decreases soil moisture and alters temperature profiles (Dong et al. 2006; Abhilasha et al. 2008; Weber 2008). Moreover, *Solidago canadensis* is known to exude potent allelopathic chemicals (e.g., polyphenolic acids, flavonoids, and essential oils) from its roots to inhibit the growth of native competitors (Judžentienė et al. 2023; Poljuha et al. 2024).

While primarily studied for effects on plant communities, these compounds may directly or indirectly affect soil organisms and microarthropods, further contributing to habitat quality decline. Moreover, due to its yellow flowers found in a large number in paniculate inflorescences (Skrzypczak et al. 1999), it attracts pollinators such as syrphids, beetles, solitary bees and bumblebees (Robson et al. 2017; Ustinova and Lysenkov 2020), through their nectar rich in carbohydrates, among which arabinose, galactose, glucose and arabinogalactan (Šutovská et al. 2013).

Although the pollen of invasive plant species can represent an additional source of food for the local pollinator communities (Kovács-Hostyánszki et al. 2022), their presence can involve a negative impact on the overspecialized pollinators and even decrease the abundance of native pollen-giving plants (Litt et al. 2024). Even though the groups of insects that pollinate this species are well diversified, the honeybees (*Apis* spp) show a significantly higher abundance (Jachuła et al. 2020). As well as pollinators, the ground-dwelling arthropods tend to respond to the invasion of *Solidago canadensis*.

Ground-beetles (Carabidae) relative abundance, species richness and activity were lower in the invaded grasslands by *S. canadensis* (Baranová et al. 2014). At least for the abundance that was also shown by De Groot et al. (2007). Yet, other ground-dwelling arthropods, such as rove (Staphylinidae) and carrion beetles (Silphidae) areas showed greater overall diversity and evenness of ground-dwelling beetles, along with a higher abundance (Baranová et al. 2014). The general impact appears to be taxon-dependent, as shown by Tóth et al. (2025), who found that certain myriapods, specifically centipedes, seem to benefit, likely because of a positive plant-soil feedback mechanism.

In the general overview above, *S. canadensis* invasion has shown contrasting results for different arthropod assemblages. Yet, there is one group that has been shown to be negatively affected by the Canadian goldenrod, the ant communities (Kajzer-Bonk et

al. 2016; Trigos-Peral et al. 2018). Ants are one of the most successful insect groups, often accounting for the largest share of biomass in terrestrial invertebrate communities (Schultheiss et al. 2022). By constantly building nests and tunnels, ants act as soil engineers. This process, known as bioturbation, turns over massive amounts of soil, significantly enhancing its quality by boosting aeration, water infiltration, and nutrient availability for plants (Frouz and Jilková 2008; Zelikova et al. 2011). Many plant species have evolved to produce seeds with attractive elaiosomes specifically for ants. The ants consume the elaiosome and leave the intact seed in a sheltered, nutrient-rich location within the nest, making them key dispersers of seeds in forests and scrublands (Beattie and Hughes 2002).

Even more, ants are highly efficient, cooperative predators, actively controlling other invertebrate populations by preying on the eggs and larvae of various pest species (Davidson et al. 2003; Rosumek et al. 2009). In some natural meadows of Poland, the spread of *S. canadensis* decreased the nests density and the ant species abundance compared to non-invaded meadows (Kajzer-Bonk et al. 2016; Trigos-Peral et al. 2018) and also changed the dominant ant species in the area (Trigos-Peral et al. 2018). The genus *Myrmica* was strongly affected by the presence of Canadian goldenrod (Kajzer-Bonk et al. 2016; Trigos-Peral et al. 2018), while other species, e.g. *Lasius niger* were stimulated in the invaded sites (Kajzer-Bonk et al. 2016; Trigos-Peral et al. 2018).

Moreover, in the case of *Myrmica rubra*, it was shown that most of their energy is used for gyne production and less for workers, as an adaptation to an environment in which the goldenrods (*S. canadensis* and *S. gigantea*) spread, to extend to a location with more advantageous conditions (Grześ et al. 2018). The spread of Canadian goldenrod was associated with a lower specific diversity of ants, and even with a reduced number of anthills, built in smaller sizes than normal, including species of the genus *Myrmica*, *Lasius* and *Formica* (Lenda et al. 2013). Ant communities were also affected indirectly by a lack of aphids on the surface of *S. canadensis* as an additional reason for ants to avoid this species (Stukalyuk et al. 2019).

This study aimed to investigate the effects of *S. canadensis* on ant communities with emphasis on the abundance, diversity and composition comparing the meadow sites invaded or non-invaded by the *Solidago canadensis* and of different site exposures in central Romania meadows.

Material and methods

Sampling protocol

Field work was carried out during August 2018 on the Târnavelor plateau (Transylvania, Romania) in six sites and twelve distinct meadows (invaded and non-invaded areas) in the vicinity of Daneș, Pelișor, Bârghiș and Biertan regions (Fig. 1). The geological substratum is composed mainly of Pannonian sedimentary deposits (clays, sands, and poorly cemented grits), as well as Sarmatian, isolated Badenian, and Quaternary deposits found along the valleys (sands, gravels of varying sizes, and mud) (Costea 2007).

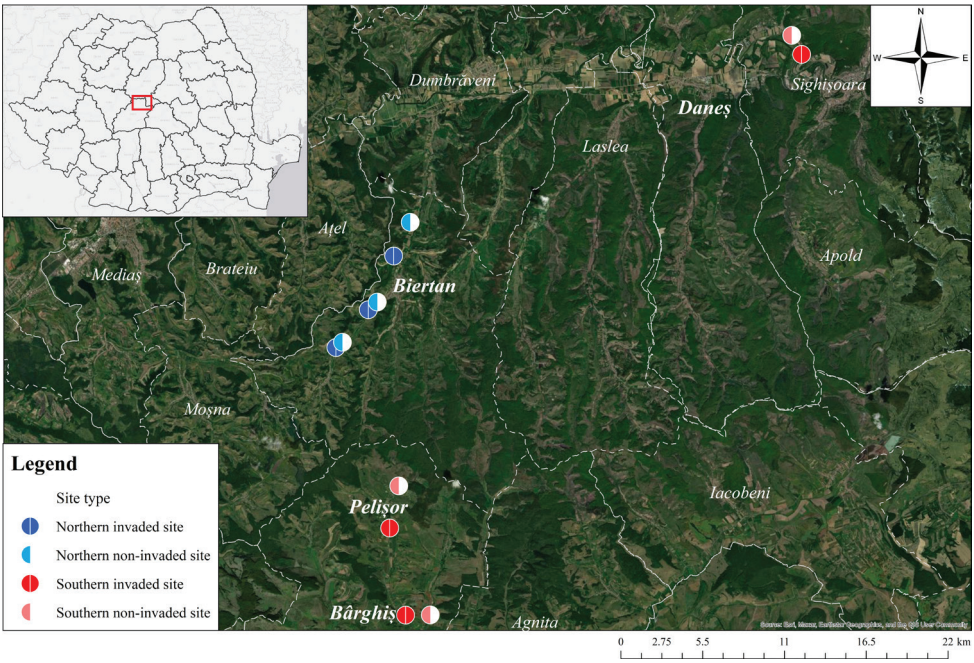


Figure 1. Sampling sites in Târnavelor Plateau, Transylvania (codes: blue circles – invaded sites with northern exposure of the sampled meadows, half blue-half white circles – non-invaded sites with northern exposure, red circles – invaded sites with southern exposure, half red-half white circles – non-invaded sites with southern exposure.

The floristic inventory of the whole plateau consists of 897 vascular taxa (843 species, 46 subspecies, 4 varieties and 4 forms), distributed in 100 families. Within this high diversity vulnerable plant species occur and form vegetation associations such as Festuco-Brometea, Rhamno-Prunetea, Querco-Fagetea (Sămărghițan and Oroian 2014).

The experimental design included two types of exposure: northern and southern. The sampling meadows in Biertan region had northern exposure, whereas those in the other investigated meadows had southern exposure. The meadows were chose based on several features such as comparable vegetation and being spatially paired, as invaded and non-invaded in each site of the six. The general landscape of the Târnavelor plateau consists of a mosaic of invaded and non-invaded patches. Yet, this landscape is varying within the area. Overall, 12 sampling meadows were investigated with 6 of them being invaded with *Solidago canadensis* and 6 were non-invaded (i.e., paired design for the effect of invasion). Out of each invaded and non-invaded three of them had northern exposure and three southern exposures (Fig. 1).

Ant sampling method

We applied a well-suited technique for collecting ground-dwelling invertebrates, the Barber pitfall traps (Brown and Matthews 2016). The glass or plastic containers of

10 cm diameter were buried in the ground so that the opening surface of the vessel was flush with the soil surface. A roof 4–6 cm higher was installed above the traps, to avoid filling them with water from precipitation. The traps were half-filled with preservative liquid (glycol – water, 1:1 solution) and collected after 14 days. A total of 180 Barber traps were installed in 12 sites. Each sampling site consisted of 15 pitfall traps in a 3 × 5 network with 5 meters distance between the traps. For sake of comparability, sampling was carried in each site where patches at least 49 m^2 were covered with at least 75% of *Solidago canadensis* for the invaded plot, and a similar surface was used for non-invaded plots, where *Solidago canadensis* was absent (Fig. 2).

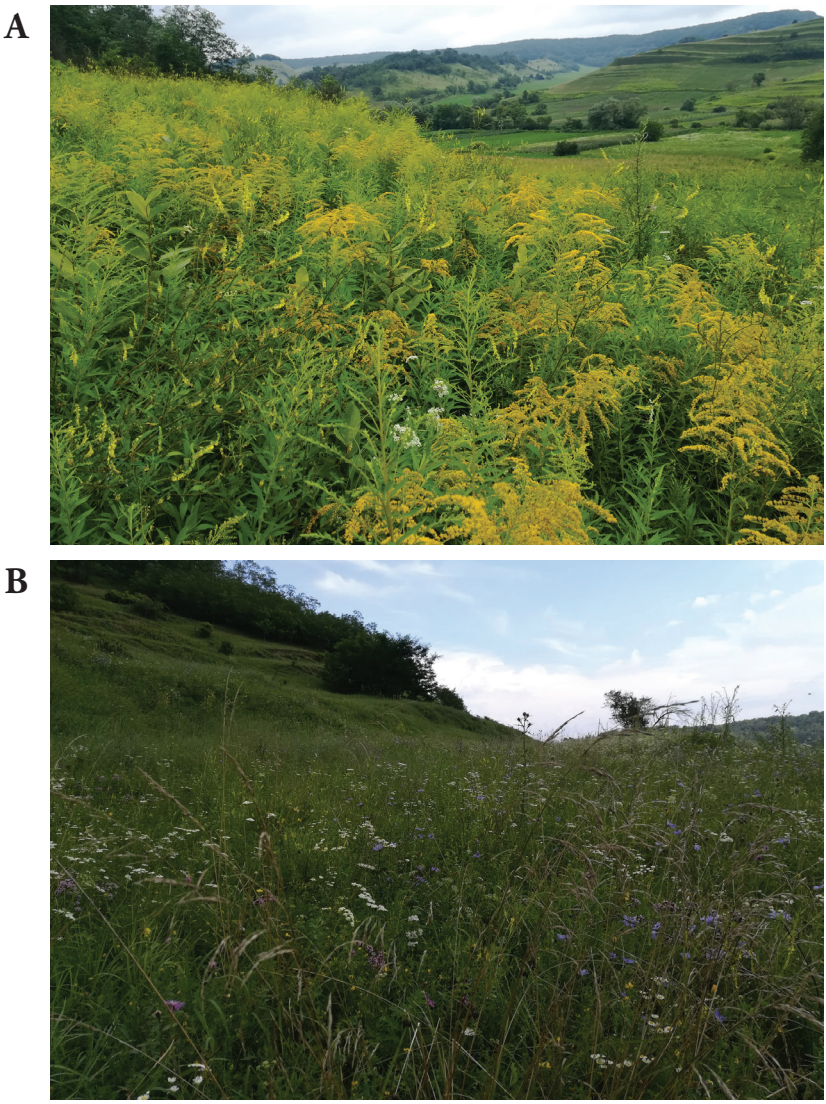


Figure 2. Sampling site with the target invasive plant *Solidago canadensis* in the front (upper part) and a non-invaded site (lower part) from Târnavelor Plateau, as an example of the habitats sampled.

Data analysis

The myrmecological material was analysed and identified to species level using the keys of Czechowski et al. (2012) and Seifert (2018). At first, we run the analysis on ant species richness and ant abundance at the level of individual pitfall traps (N = 162). Due to some pitfall destruction, we retrieved 162 of originally placed 180 traps and the statistical analysis was based on this data. Normality was tested using Shapiro-Wilk Test. The GLMM analysis was employed to test the main and interactive effects of exposure and invaded/non-invaded status on ant abundance and ant species richness in each trap, while specifically controlling for the non-independence of samples collected from each site. The model included the exposure and invaded/non-invaded status and their two-way interaction as fixed effects whereas the sampled sites (N = 6) were included as a random intercept. This structure controls for the spatial clustering of observations within sites. The model was fitted with a Poisson error distribution and a log link function. The model's residuals were examined for overdispersion which was calculated as the sum of squared Pearson residuals divided by the residual degrees of freedom.

Next, diversity analysis in terms of species richness and Shannon–Weiner index, and multivariate analysis (PCA, RDA) were carried out at the level of sampled meadows (N = 12). To account for differences in sampling effort caused by the loss of pitfall traps (see above), we standardized species richness comparisons using sample-size-based rarefaction and extrapolation curves to 15 traps (samples) in each meadow (Chao et al. 2014). Analyses were performed using the iNEXT package (Hsieh et al. 2016) in R (see Suppl. material 1: table S1). Species abundance data were then aggregated (summed) by creating a unique sampling unit, defined by the combination of the sample site, site type (Invaded/Non-Invaded), and exposure (Northern/Southern). The iNEXT function was applied to the resulting assemblage abundance data to estimate asymptotic diversity metrics. The order of diversity q was set to q=0 (S_0 = Species Richness) and q=1 (S_1=Exponential Shannon Index) which measures the number of typical species. To test the effects of the environmental factors on the estimated diversity metrics, two separate Linear Mixed Models (LMMs) were constructed using the estimated diversity values as the response variables. The significance of the fixed effects (Invaded/Non-invaded status, exposure, and their interaction) was assessed using Type III ANOVA**.** The six sites were again put as a random factor to reflect the paired design**.**

Principal Component Analysis (PCA) was performed on the Hellinger-transformed species abundance matrix to reduce dimensionality and visualize the primary gradients of variation in the ant community structure. Data was analysed using R software and RStudio (R CoreTeam 2020) (packages: dplyr, ggpubr, ggplot2, vegan, iNEXT, car, decostand). Furthermore, we run the Redundancy Analysis (RDA), a constrained ordination, to test the hypothesis, if the variance in the ant community species composition could be explained by the explanatory variables (i.e., exposure and invaded/non-invaded status). The model was constrained by these two variables as fixed factors along with their two-way interaction following the Hellinger transformation of the species data. To account for known spatial and environmental differences

among the sampling locations that might obscure the effects of the primary factors, the variable site (the sampled six sites) was included as a covariate. The latter analyses were performed using the Canoco software (ter Braak and Smilauer 2012).

Results

General overview

Altogether, we identified 21 species belonging to four subfamilies (see details in Suppl. material 1: table S2). As expected, all the species are common in Romania (Markó et al. 2006). In terms of zoogeographical elements, the Euro-West-Siberian and South-Palearctic species were found dominant, whose specific habitats are dry meadows and dry habitats (both open and forests).

Ant species richness analysis at *S. canadensis*-invaded and non-invaded sites at the level of individual traps

Comparing the ant species number per trap between the invaded and the non-invaded habitats we found no significant differences (Table 1, $p>0.05$), yet a slightly higher number of species was observed in the non-invaded sites (Fig. 3). The similar pattern was shown in the Northern and Southern sites, with more species in the Northern sites, yet no significant difference was found (Table 1, $p>0.05$) (Fig. 3). The interaction effect of exposure and invasion was also not significant ($p = 0.533$, Table 1). Similarly, neither the main effect of invasion status ($p=0.61$) nor exposure ($p=0.97$) significantly influenced species richness per trap (see Table 1).

Ant abundance analysis at *S. canadensis*-invaded and non-invaded sites at the level of individual traps

We found no statistically significant main effect of invasion status on total ant abundance per trap (GLMM, $z = 1.46$, $p = 0.144$), indicating that *S. canadensis* invasion did not reduce ant abundance across all sites at the fine scale, although there was a trend of slightly increased ant abundances in the non-invaded sites (Fig. 4). There was

Table 1. Results of the Generalized Linear Mixed Model (GLMM) analysis testing the effects of Invasion Status (*S. canadensis* invaded vs. non-invaded) and Exposure (Northern vs. Southern) on ant species richness per trap.

Factor	Estimate	Std. Error	z-value	p-value
Intercept (Northern, Invaded)	2.57	0.47	5.52	**< 0.001**
Invasion (Non-Invaded)	0.34	0.67	0.51	0.609
Exposure (Southern)	-0.02	0.67	-0.03	0.973
Invasion × Exposure	-0.59	0.94	-0.62	0.533

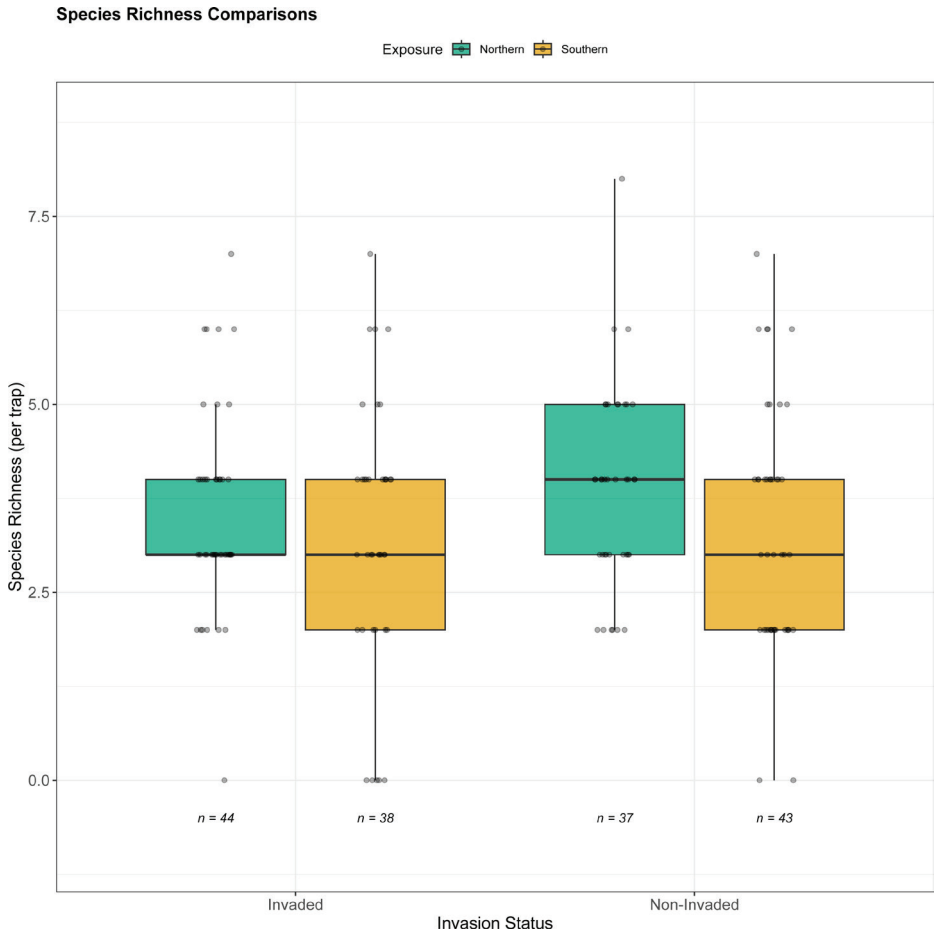


Figure 3. The variation in ant species richness per pitfall trap within the *Solidago canadensis*-invaded and non-invaded sites with northern and southern exposure. For associated statistical tests using GLMM, see Table 1.

Table 2. Results of the Generalized Linear Mixed Model (GLMM) analysis testing the effects of Invasion Status (*S. canadensis* invaded vs. non-invaded) and Exposure (Northern vs. Southern) on ant abundance per trap.

Fixed effect	Estimate	Std. Error	z-value	p-value
Intercept (Northern, Invaded)	3.29	0.40	8.18	**< 0.001**
Invasion (Non-Invaded)	0.67	0.46	1.46	0.144
Exposure (Southern)	-1.09	0.58	-1.89	0.059
Exposure (Southern)	-1.09	0.58	-1.89	0.059

a marginally significant effect of exposure (GLMM, z = -1.89, p = 0.059), with a trend towards lower abundance on southern slopes compared to northern slopes (Table 2, Fig. 4). The interaction between invasion status and exposure was not significant (z = 0.45, p = 0.651). The random effects analysis showed considerable spatial variation in

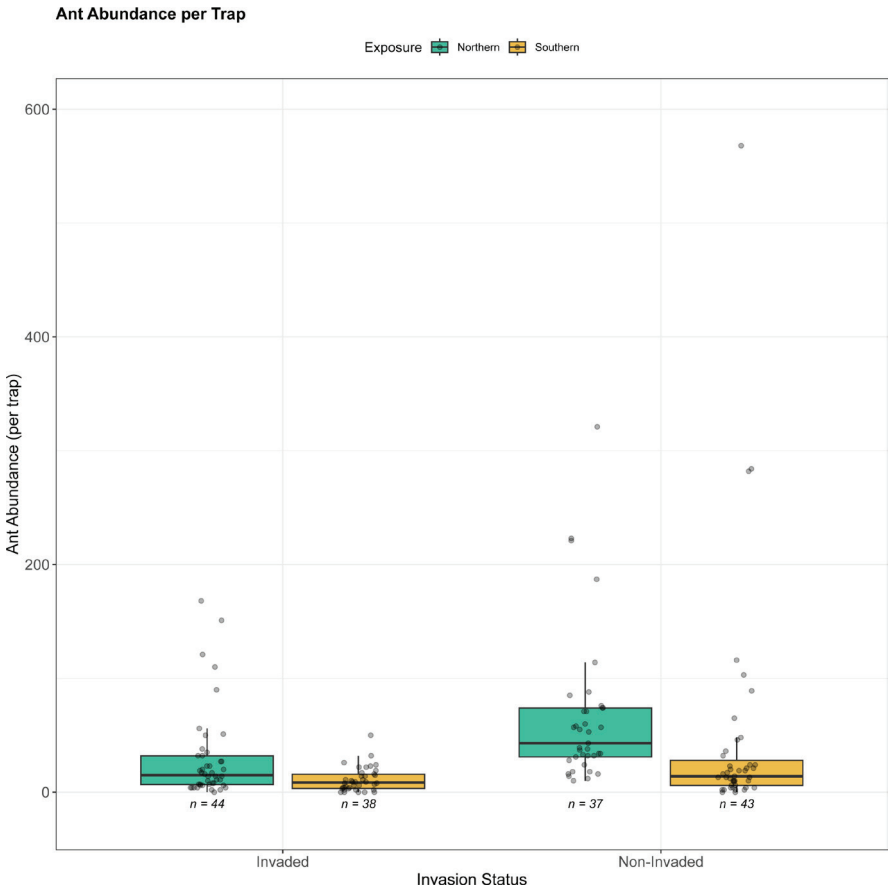


Figure 4. The ant abundance variation per pitfall trap within the *Solidago canadensis*-invaded and non-invaded sites with northern and southern exposure. For associated statistical tests using GLMM, see Table 2.

abundance between sites (Variance = 0.18) and meadows (Variance = 0.23) (Table 2). Interestingly, the northern exposure was characterized with higher abundances compared to the southern exposure.

Analysis of ant diversity at *S. canadensis*-invaded and non-invaded meadows

The LMM (Linear Mix models) for the Estimated Species Richness (S_0 – Estimated Richness) revealed that neither the main effects nor the interaction significantly affected the number of species present. The random intercept for the sites showed a relatively large standard deviation, indicating that there were considerable baseline differences in species richness among the study sites ($p>0.05$, Table 3). Although the estimated species richness of ant communities did not differ significantly based on the presence of invasive species or the exposure of the site, probably due to too small number of replicates and that large variance, the median species richness per meadow was about twice as high on northern slopes as on southern slopes (Fig. 5).

Table 3. Linear Model (LM) slope estimates testing the main and interactive effects of Invasion Status (*Solidago canadensis* invaded vs. non-invaded) and Exposure (Northern vs. Southern) on Estimated Species Richness (S_0) per site.

Factor	Estimate	Std. Error	t-value	p-value
Intercept (Northern, Invaded)	14.99	2.72	5.52	**< 0.001**
Invasion Status (Non-Invaded)	0.00	3.84	0.00	1.000
Exposure (Southern)	-5.66	3.84	-1.47	0.179
Invasion × Exposure	0.00	5.43	0.00	1.000

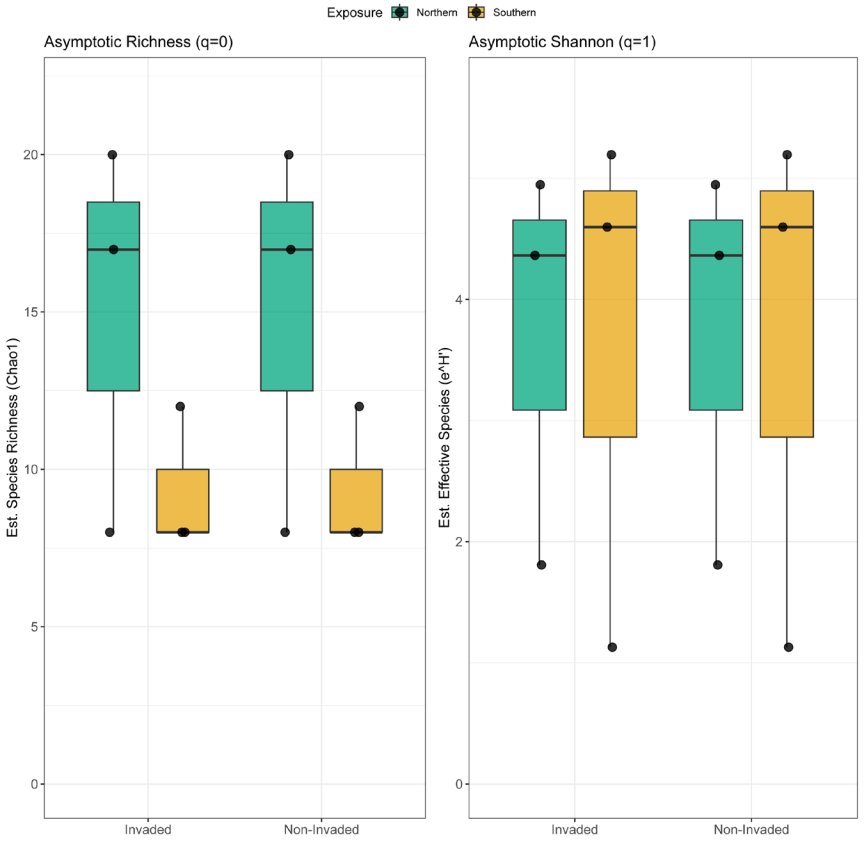


Figure 5. Asymptotic diversity profiles (q=0 and q=1) by site type and exposure at meadow level. For associated statistical tests using LMM, see Tables 3, 4.

The Linear Mixed Model (LMM) for the Exponential Shannon Index (S_1) showed also no significant main effects of invasion status (t = 0.15, p = 0.887) or exposure (t = 0.66, p = 0.528) and the variance due to the exposure was much lower, suggesting that only rare species contributed to the increased species richness on the northern slope (Fig. 5). Contrary to expectations, the interaction between invasion status and exposure was also not significant (t = -0.55, p = 0.595, Table 4, Fig. 5).

Table 4. Results of the linear mixed-effects model (LMM) testing the effects of Invasion (Invaded vs. Non-Invaded) and Exposure (Northern vs. Southern) on the Estimated Exponential Shannon Index (S1).

Factor	Estimate	Std. Error	t-value	p-value
Intercept (Northern, Invaded)	6.80	0.97	6.98	**< 0.001**
Invasion (Non-Invaded)	-0.95	1.38	-0.69	0.510
Exposure (Southern)	-0.93	1.38	-0.67	0.521
Invasion × Exposure	0.62	1.95	0.32	0.760

Ant community composition at *S. canadensis*-invaded and non-invaded meadows

The results of the PCA analysis regarding the ant assemblage composition related to the investigated sites are presented in Fig. 6. The first two axes of the PCA plot explain approximately 48.6% of the variability in Formicidae species composition. Within these axes, the species *Lasius niger*, *Formica cunicularia*, *Lasius alienus*, *Formica rufibarbis* occur mainly in both invaded and non-invaded habitats but with a preference for Northern exposure, whereas species such as *Myrmica scabrinodis*, *Solenopsis fugax*, *Myrmica rubra* and *Lasius flavus* occur mainly in Southern-exposed meadows which are

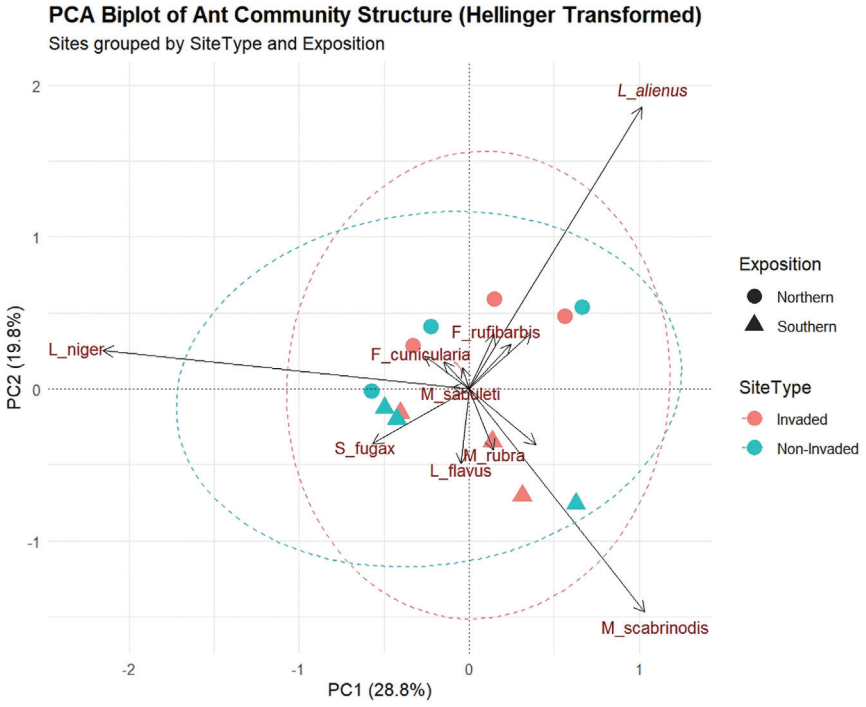


Figure 6. PCA analysis of the ant communities in relation with the investigated sites using Hellinger Transformed – Eigenvalues: Axis 1 – 0.18114, Axis 2 – 0.1632. Explained variation (cumulative) Axis 1 – 0.3524, Axis 2 – 0.6701; codes for species -first letter of the genus, followed by the name of the species; red circle – invaded sites, blue circle – non-invaded sites, black triangle – sites with southern exposure, black circle -sites with northern exposure.

both invaded and non-invaded. However, *Myrmica sabuleti* and *Formica clara* showed less preferences towards the exposure status of the site (Fig. 6). Overall, the effect of invasion was not evident in the PCA analysis (invaded and non-invaded sites are mixed up over the space of the diagram).

In the constrained RDA analysis, we observed that the exposure explained a 26.8% of the observed variation in the composition of the ant communities and correlated with the first axis (Exposure.N, $p = 0.004$; Exposure.S, $p = 0.013$). A clear delimitation is observed in two distinct groups of species depending on the exposure. The species *Lasius alienus*, *Polyergus rufescens*, *Formica rufibarbis*, *Camponotus aethiops*, *Camponotus piceus* and *Myrmica gallienii* were predominantly found in habitats with northern exposure, such as those in Biertan sites, while *Tapinoma erraticum*, *Lasius flavus*, *Solenopsis fugax*, *Myrmica scabrinodis*, *Ponera coarctata* and *Myrmica sabuleti* were more frequently found in habitats with southern exposure, as are those of Daneș, Pelișor and Bârghiș. In contrast, the invasion had against our expectation no significant effect on the species composition, although there was an indication that *Tetramorium* cf. *caespitum* increased in the most of invaded sites (Fig. 7, Table 5, p = 0.936).

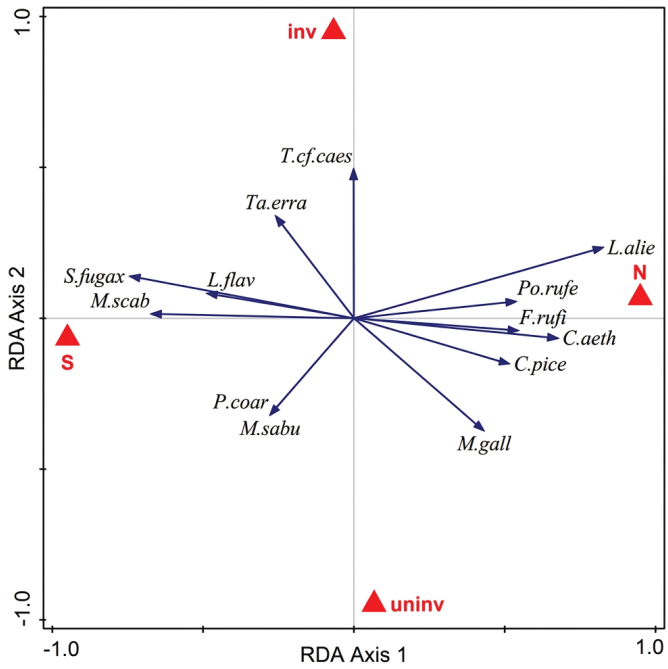


Figure 7. RDA analysis (step 'Forward Selection') of the ant communities in regards the invasive /non-invasive character of the sites, and the exposure— S – southern exposure, N – northern exposure, uninv – non-invaded, inv – invaded; codes for species: *L. alie* – *Lasius alienus*, *Po. rufe* – *Polyergus rufescens*, *F. rufi* – *Formica rufibarbis*, *C. aeth* – *Camponotus aethiops*, *C. pice* – *Camponotus piceus*, *M. gall* – *Myrmica gallienii*, *T.* cf. *caes* – *Tetramorium* cf. *caespitum*, *Ta. erra* – *Tapinoma erraticum*, *L. flav* – *Lasius flavus*, *S. fugax* – *Solenopsis fugax*, *M. scab* – *Myrmica scabrinodis*, *P. coar* – *Ponera coarctata*, *M. sabu* – *Myrmica sabuleti*. For significance of predictors see Table 5.

Table 5. RDA statistical summary regarding the factors that explain ant community composition variations.

Forward Selection Results:					
Name	**Explains %**	**Contribution %**	**pseudo-*F***	***p***	***p*(adj)**
N exposure	26.8	87.1	3.7	*0.004	**0.008**
S exposure	26.8	87.1	3.7	*0.013	**0.013**
Invaded/Non-invaded	4	12.9	0.5	0.936	0.936

Discussions

The invasion of alien goldenrods (*Solidago gigantea* and *S. canadensis*, Asteraceae) is considered to be one of the major threats for semi-natural habitats in Central Europe (EPPO 2024; Kajzer-Bonk et al. 2016).

For instance for the ants, *S. canadensis* decreased the ant nest densities by more than half in invaded areas in Poland (Kajzer-Bonk et al. 2016). The reduction of ant abundance was observed also by Trigos-Peral et al. (2018). Due to the invasive character of *S. canadensis*, the invaded habitats were characterized by lower moisture which negatively influenced the abundance of *Myrmica* species in the invaded meadows (Trigos-Peral et al. 2018). However, in our study we observed that the ant abundance was not significantly different regardless of the invasion status. Yet, the northern exposed meadows were preferred by ants, at least in terms of abundance (slightly) and species composition (the largest effect in our study). The management factor can be an answer here. All the sites with northern exposure in our study (Biertan region) are managed by traditional mowing at least two times per year. Thus, the structure of vegetation can shift and may enhance abundance of ants in these habitats, compared to less frequently mowed habitats. Moreover, other studies have shown that the highest positive effect on the re-establishment of native plant species was recorded with two mowing events per year in meadows invaded by *Solidago* sp. (Hall et al. 2022).

In terms of diversity, studies have shown a decrease of species richness in the plots where *S. canadensis* occurred. The average number of ant species per plot decreased from 3.03 in control areas to 2.3 in goldenrod-invaded plots (Kajzer-Bonk et al. 2016). Similar results were obtained by Trigos-Peral et al. (2018). Moreover, Grześ et al. (2018) have shown that besides diversity, the colonies production may be also affected. Yet, in our study we observed no significant differences between the invaded and non-invaded meadows. There are two hypotheses that can be further tested. The overall landscape, which is a mosaic structure of invaded and non-invaded meadows has been affected by *S. canadensis* invasion during decades and uniformized the meadows, or the success of control measures may be strongly related to site-specific conditions and environmental factors, implying that ant responses to *Solidago* invasion could be too variable (Hall et al. 2022).

The ant assemblages showed intriguing features in terms of species individual responses. For instance, *Lasius alienus* and *Solenopsis fugax* were detected in larger numbers in south-exposed invaded habitats, whereas *Myrmica gallienii* preferred the north-exposed uninvaded meadows. Although the overall response to the invasion itself by the species were not significant surprisingly in our study, as the effects of the slope, and

perhaps also of the individual sites, override that of invasion itself. Similar to the study of Hall et al. (2022), environmental factors may explain the obtained results. Slope aspect is a well-established driver of microclimatic variation in temperate ecosystems, with southern exposures typically experiencing higher solar insolation and soil temperatures compared to northern slopes (Cantlon 1953; Bennie et al. 2008). However, in our study, this expected thermal advantage did not translate into significant differences in ant abundance, possibly due to vegetative buffering or high intra-meadow heterogeneity. Moreover, in harsh environments (such as invaded habitats), cooler and more humid north-facing slopes may act as microrefugia. These shaded areas protect ants from lethal temperatures and desiccation. Because of this, they often support unique, functionally specific ant assemblages that simply could not survive the harsh, xeric conditions of intensive heat typical southern slopes (Bátori et al. 2020). Studies show that topography, such as north-facing slopes and deep forest covers, fosters highly distinct ant communities that differ completely from those on warmer, flatter, or south-facing terrain (Davis and Zigler 2012). We confess, however, since our sampled meadows at Northern slope were all situated in the Biertan region with more frequent mowing, we can not distinguish the effects of the management from the regional microhabitat's effects in our study. Yet, it is likely both these factors play an important role for the ants, unlike the invasive plant we studied.

Conclusions

Despite much evidence that invasive species have a general negative effect on native species, our results show that in the case of ant species richness and abundance is not affected in the meadows in central Romania. Notably, although there are clear differences in terms of species composition between meadows of northern and southern exposure, no such significant differences were observed between the invaded and non-invaded sites, and the observed changes are so driven rather by the regional and/or microclimatic variance. Our results show that the invasive plant species *Solidago canadensis* causes rather a limited negative effect on diversity and abundance of the ant communities in Romania, while other parameters (e.g. environmental factors, the overall landscape) must be included in the community ecology analysis.

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Suppl. material 1

Supplementary table

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Data type: docx

Explanation note: **table S1.** Sampling effort, observed ant species richness, and sample-size-based extrapolated richness (standardized to 15 traps) across the 12 meadows. **table S2.** Zoogeographical and ecological classification of ant species.

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