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Using private gardens to conserve *Minuartia smejkalii* - a novel approach for *ex situ* conservation

Solveig Franziska Bucher [a,b,c,d,*], Zuzana Münzbergová [d,e], Hana Pánková [d]

[a] *Institute of Biodiversity, Ecology and Evolution, Department of Plant Biodiversity, Friedrich Schiller University Jena, Jena, Germany*
[b] *Senckenberg Institute for Plant Form and Function (SIP) Jena, Jena, Germany*
[c] *German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Leipzig, Germany*
[d] *Institute of Botany, Czech Academy of Science, Průhonice, Czech Republic*
[e] *Department of Botany, Faculty of Science, Charles University, Praha, Czech Republic*

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ABSTRACT

We currently observe a tremendous loss of species due to the ongoing global change. *Ex situ* conservation provides an important tool to counteract species losses and thus helps to maintain biodiversity. Here, we present a setup which involves 19 private gardens and 2 botanical gardens as sites of *ex situ* conservation using *Minuartia smejkalii*, an endangered serpentinophyte endemic to Czechia as a case study. To create suitable habitats in the gardens, artificial serpentine mounds were built, combining soil from natural sites with serpentine rocks and gravel from nearby quarries. We investigated how the plants performed at *ex situ* conservation sites compared with plants growing in natural habitats and how well these artificial sites represented the conditions at the natural sites. We assessed plant performance using plant functional traits including tuft diameter, chlorophyll fluorescence, chlorophyll content, and specific leaf area (SLA). We found that even though abiotic factors of the *ex situ* conservation sites (temperature, soil properties and shading) did not entirely match the natural conditions and plant mortality was higher and performance lower under *ex situ* conditions, we managed to create many independent *M. smejkalii* populations and increase its variability in plant functional traits, which might facilitate its survival. Using private gardens as *ex situ* conservation sites also helped to raise awareness on the conservation of *M. smejkalii* through engagement of the local community and the gardeners and thus, this proves to be a useful and powerful technique in plant conservation.

1. Introduction

Life on earth is currently facing major changes, as due to the ongoing global change, up to 40 % of the global plant diversity face extinction through the loss, fragmentation, or deterioration of suitable habitats (Fahrig, 2003; IPCC, 2007; Lindenmayer and Fischer, 2013; Nic Lughadha et al., 2020). *Ex situ* conservation is an effective way of conserving highly endangered plants in man-made facilities such as botanic gardens or arboreta which help safeguard vital plant material for reintroduction into wild stands (Raven, 2004; Mounce et al., 2017; Abeli et al., 2020).

While cultivation in *ex situ* living collections seems an elegant, easy to apply approach to species conservation, it also has a lot of problems. Habitat conditions of *ex situ* sites are often very artificial due to the garden management and very different from the original conditions. Most of the sites are located far away from the natural occurrence of the species and the plants are thus experiencing very different conditions. The facilities are often linked to university or institutional gardens in big cities, which makes genetic exchange between natural populations and *ex situ* sites virtually impossible but increases the risk of hybridisation due to the vicinity of many other related plant species (Stojanova et al., 2021; Lozada-Gobilard et al., 2020). Botanical gardens are also restricted in their available space which makes it difficult to foster genetically divers populations there rather than just planting a few individuals (Ismail et al., 2021). Additionally, if the plants occur in more than just one natural population, seeds from different origins within the same plant species should be kept separate and not planted together in an *ex situ* conservation site to maintain genetic distinctiveness (Stojanova et al., 2020).

Plants in *ex situ* collections may also adapt to the novel conditions within the gardens and thus become maladaptated to their native sites,

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* Corresponding author at: Philosophenweg 16, 07743 Jena, Germany.
E-mail address: Solveig.franziska.bucher@uni-jena.de (S.F. Bucher).

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which makes them unsuitable for reintroduction into the wild (Enßlin et al., 2011; Ensslin and Godefroid, 2019; Ensslin et al., 2023). Previous research showed that plants from *ex situ* cultivation sites often suffer from unfavourable changes in life history traits, reduced plant performance or loss of genetic diversity due to the low sizes of the population which are maintained and selective sampling of the individuals which are conserved (Raven, 2004; Ensslin et al., 2015; Ensslin and Godefroid, 2019, 2020; Lozada-Gobilard et al., 2020; Stojanova et al., 2020; Stojanova et al., 2021).

Here, we propose to use private gardens, *i.e.*, gardens associated with houses or schools which are maintained by lay-people in the vicinity of natural, endangered populations for *ex situ* conservation as a promising and novel approach for *ex situ* conservation. Private gardens are very suitable as they outnumber botanical gardens by several orders of magnitude. Their use in urban conservation gardening has been proposed by Segar et al. (2022) who suggested the cultivation of declining native plant species in public and private green spaces and the subsequent adjustments necessary in the horticultural market. Munschek et al. (2023) used Germany as a case study and designed an app to make recommendations to gardeners of region-specific lists of plants including recommendations for planting and purchasing based on Red List information. There are comparably few costs associated, as plants are looked after and cared for by volunteer garden owners. The gardens are ideally located close to the natural populations, making the conditions such as climate, bedrock as well as biotic interaction more similar to natural conditions than far-away botanical gardens. Having multiple replicates of *ex situ* sites rather than just a few also increases the variability of the sites and thus foster diversity in the plant populations. The local gardens can thus also serve as steppingstones for natural (re)colonization of the region in some cases. This has been partially shown by Staude (2024b) who suggested that the cultivation of native plants in gardens has a positive impact on their occurrence in the wild. Using private gardens and engaging the local communities also help to raise public awareness to the focal species as well as conservation in general (Lin et al., 2018).

In this case study, we focussed on *Minuartia smejkalii* Dvořáková, an endemic serpentinophyte native to Czechia (Dvořáková, 1988; Pánková, 2021). To establish the plant species in private gardens, we created artificial mounds on serpentine substrate in 19 private gardens and 2 botanical gardens. To evaluate the success of this conservation measure, we studied the performance of the plants. We analysed plants grown at the original locations and at *ex situ* conservation sites in private gardens to judge whether plant performance was affected by the different environments. We assessed plant performance *via* plant functional traits linked to ecophysiology of the species and selected methods which pose a minimum disturbance for the individual plants as the number of individuals was very low (Violle et al., 2012; Pérez-Harguindeguy et al., 2013; Bucher et al., 2018). More specifically, we studied the plant diameter of the individuals, which indicates their competitive ability and vigour (Pérez-Harguindeguy et al., 2013) and used chlorophyll fluorescence methods to monitor plant stress levels and performance. As chlorophyll fluorescence parameters, we assessed the F_v/F_m, indicating plant stress, *i.e.*, the proportion of photons used in photochemistry, as well as the performance index PI_{abs}, which indicates plant performance (Clark et al., 2000; Strasser et al., 2000; Bucher et al., 2018). In addition to that, we measured the chlorophyll content as it scales well with photosynthetic performance (Evans, 1989; Zhu et al., 2012) and the specific leaf area, as it indicates resource use and growth rates (Hulshof et al., 2013; Pérez-Harguindeguy et al., 2013).

We asked the following questions:

1. How well do the artificial mounds created in private and botanical gardens capture the abiotic conditions of the natural habitats?
2. Does plant performance, assessed *via* plant functional traits, differ between *ex situ* conservation sites and natural populations?
3. To what extent do the differences in abiotic conditions affect plant performance?

With this research, we explore the role of private gardens as sites for *ex situ* conservation. Using private gardens for *ex situ* conservation will help to maintain biodiversity by both raising public awareness and creating suitable, new, and multiple habitats for endangered species to persist in. The gardens might also serve as steppingstones for further recolonisation and expansion of the populations of endangered species in nature.

2. Material and methods

Minuartia smejkalii is a herbaceous perennial with a long life span (appr. 15–20 years). It belongs to the *Minuartia verna* agg. complex, which was distributed widely during the last glacial maximum and survived and differentiated in interglacial refugia on extreme substrates (Dvořáková, 1988; Stojanova et al., 2020). Nowadays, *M. smejkalii* only occurs in two regions (Hadce u Želivky National Natural Monument and Hadce u Hrnčíř Natural Monument), which are appr. 30 km apart, and is thus critically endangered (Grulich, 2012; Lozada-Gobilard et al., 2020; Stojanova et al., 2020). We accounted for that in our analysis, as *M. smejkalii* was shown to have high genetic diversity within populations, low to moderate differentiation between populations within region and moderate regional differences (Stojanova et al., 2020; Zhu et al., 2021). In former times, the species existed in a third region, Borecká Skalka Natural Monument, but this population has gone extinct due to mining activities (Stojanova et al., 2020). The fragmentation of the population is in part due to the patchy occurrence of serpentine soils, on which the species abides. Serpentine soils are characterized by high concentrations of toxic heavy metals (magnesium, nickel, cobalt, chromium), low levels of nutrients (potassium, phosphorus, nitrogen), and are often steep, rocky, and subject to erosion. They display low water retention, providing extreme conditions for plants (Brady et al., 2005; Stojanova et al., 2020). Before the start of the conservation activity, the population size of *M. smejkalii* in the two remaining natural sites had been decreasing rapidly (from 1252 individuals in 2011 to 434 individuals in 2015).

The main threats to the remaining populations in the wild are human-mediated fragmentation such as the creation of denser forests and construction activities such as a highway and a dam. Particular populations are endangered by high competition by surrounding vegetation, intensive grazing of tufts by wild animals such as fallow deer, deer or hare, illegal waste deposition or plant collection, as well as vandalism (Stojanova et al., 2020; Pánková, 2021). We thus initiated a conservation action to help the plants both surviving in their natural locations and thriving in multiple, nearby locations in private gardens. We studied plants in 19 private gardens representing novel *ex situ* conservation sites (10 for plants from Hrčíře and 9 for Želivka), two botanical gardens (one for each natural site) and the two remaining natural populations where *M. smejkalii* still occurs in the wild (hereafter referred to as 'natural populations'; Fig. 1). The two *ex situ* conservation sites, private gardens and botanical gardens are hereafter referred to as 'gardens'. As the natural populations were big and patchy, two sites within each natural population were selected to account for their variability. Thus, in total, 25 sites were studied.

To assess the vitality of plant populations, we analysed plant functional traits of the plants and monitored the abiotic site conditions in the wild as well as in the *ex situ* conservation sites (gardens as well as botanical gardens) to see, whether the creation of artificial habitats in the gardens mimicking the natural sites was successful. Measurements were carried out in July and August 2021.

2.1. Establishment of ex situ conservation sites

Ex situ populations were planted from propagated material in 2017–2020. Seeds were collected in both remaining natural sites: from the whole population in Hrnčíře and from the largest site in Želivka. In both populations we collected seeds from most of the flowering plants

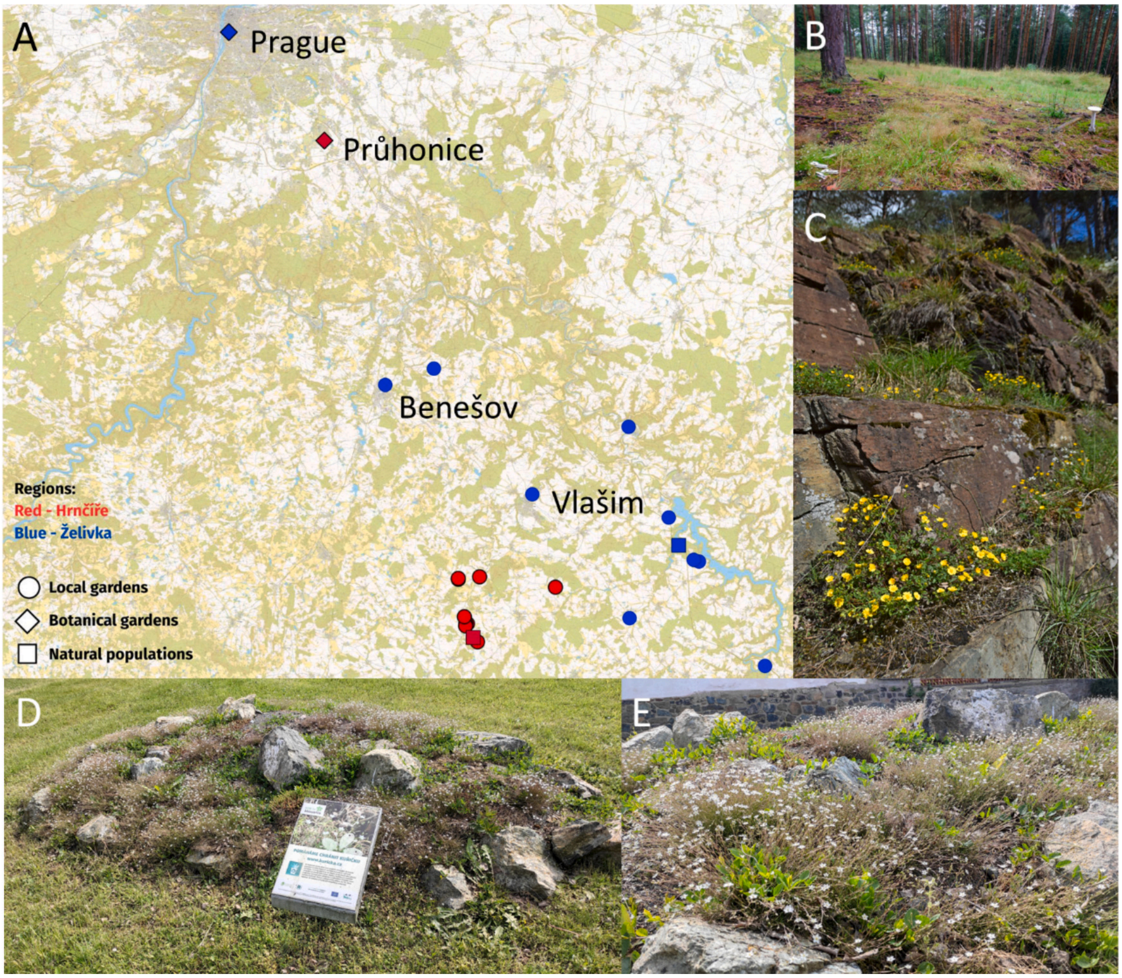

Fig. 1. A) a map of the natural populations and the *ex situ* conservation sites. B) plants at the natural population in Hrnčíře, C) plants at the natural population in Želivka, C) and D) plants at an *ex situ* conservation site on a village square and in a private garden.

(up to 5 % of the current production per plant) in 2016–2018 across the whole natural population. In total, seeds were collected from 60 individuals in Želivka and 45 individuals in Hrnčíře. The total seed production was very high - the average number of seeds produced per individual was 1864 seeds per year (Hrušková, 2021). Seeds were geminated directly on serpentine soils and the juvenile plants were also grown on serpentine soils to maintain and foster the adaptations to natural conditions. A part of the seeds harvested was germinated in Petri dishes to evaluate the germination rate.

In total, 1035 juvenile plants were planted in private gardens and 508 in botanical gardens. Each garden received 15–358 individuals depending on the area available in the garden. Plants from the two regions were kept separately which was also accounted for in the analysis to see, whether there is a difference between plants from the two regions. The distribution of plants from both populations was based on their distance from the natural sites, *i.e.*, gardeners closer to Želivka obtained individuals from Želivka. The only exception were the two botanical gardens, which are far from both natural populations (around 75 km) and thus one of them obtained seeds from Želivka and the other from Hrnčíře.

As the species is a serpentine specialist, serpentine mounds were created in each garden consisting of soil from the original locations to create the native vegetation cover from seed bank and serpentine rocks from a nearby quarry to provide optimal conditions for the plants. The extent of the mounds ranged between 1.5 to 40 m^2, and 0.1 to 1 m in height depending on the capacity of each garden. Environmental conditions were different between gardens, but most of the mounds were built in open areas as the gardens did not have large trees. Therefore, the environmental conditions were closer to the situation of the natural population in Želivka, which is a very open forest than in Hrnčíře, where the forest is much denser.

The gardeners were instructed how to care about the plants (low watering, no addition of nutrients, weeding only of non-serpentine plants). Plant survival and growth is still evaluated every year starting from the establishment of the *ex situ* cultivation up to now. All newly established plants obtained identifying numbers, and their growth is evaluated. Parts of the seeds from original as well as seeds from newly established plants were collected and their germination evaluated. Generally, the survival rate of transplanted plants was high in the first year (73 % on average). It, however, dropped to 25 % in the following year. The collected seeds had high germination rate (70–95 %, pers. obs.). All *ex situ* populations were thus able to produce offspring and, thus, the populations were maintained naturally at the *ex situ* conservation sites (Pánková, 2021).

Local and botanical gardens are not distinguished in the results since the methodology of creation of mounds as well as the instructions for plant care were the same. For our study, we analysed the offspring originated in the gardens, not the originally transplanted individuals, to avoid any effect of transplantation.

2.2. *Abiotic site conditions*

Alongside the measurements of plant functional traits, we measured local site conditions to see how the artificial mounds differed from natural sites and how characteristics of the gardens affected the performance of the plants. We placed temperature sensors to measure temperatures above the soil (+15 cm respective to ground level), on soil surface (+2 cm), and belowground (6 cm below ground) to determine the temperature conditions the plants encounter, as well as soil moisture in selected locations (TMS-4, TOMST, Prague, Czechia; Wild et al., 2019). As the individuals of *M. smejkalii* were planted in small artificial mounds in the gardens, we measured temperature and moisture on top of the heaps, as well as on the sides to capture the variability within the mounds. The same was done for natural populations, where we selected 3 sites which showed the largest differences in terms of canopy openness. The sensors logged temperatures and moisture every 15 min. The temperature sensors logged the data in between July 31, 2021, and June 23, 2022.

We also measured the leaf area index (LAI) to assess the shading (LAI-2200, Li-Cor Bioscience, Lincoln, Nebraska) of all populations *via* measuring the spectral composition of the light just above the vegetation and below the vegetation at soil level and analysed soil chemical composition for each mound. We measured the pH of the soil, N and C content (%) as well as Mg and P (mg/kg). Soil samples were sieved using a 2 mm sieve and processed as described in Pánková et al. (2008). One soil sample was lost and therefore not included in the analyses.

2.3. Functional trait measurements

At each site, ten green, vital newly established individuals were selected (if available). For the artificial mounds in the gardens, we collected samples from plants on top of the mounds and the base of the mounds to assess variability within the gardens, since we could see strong differences between the two locations. Plants located on top of artificial mounds in gardens were much less vital than at the edges, where they were shaded and presumably moisture levels were higher. As most of the plants on top of the mounds were dried out above ground due to the warm temperatures, they could not even be measured for plant functional traits.

The leaves of *M. smejkalii* are very small, thus, the measurements were carried out on entire leafy stems as the branches are non-woody and thus do photosynthesize. For each individual selected, we measured plant dimensions namely the maximum plant width (diameter), a measure for competitive strength (Pérez-Harguindeguy et al., 2013). We analysed the chlorophyll fluorescence to get information about stress physiology using a PocketPEA device (Hansatech Instruments, King's Lynn, UK) in the field. With this device, the two parameters F_v/F_m, a measure of stress and PI_{abs}, a measure of the photosynthetic performance of the individuals were recorded after 30 min of dark adaption of the leaves (Strasser and Srivastava, 1995; Clark et al., 2000; Strasser et al., 2000; Bucher et al., 2018). Subsequently, we detached and transferred some leafy stems to the lab, storing and transporting them in moist plastic bags. For each individual, we measured chlorophyll content with an atLeaf PLUS device in the lab (atLeaf, Wilmington, Delaware, USA), which is closely correlated with maximum photosynthesis rates (Zhu et al., 2012). Due to the small size of the leafy stems, this was not possible for all samples. We also scanned the leafy stems with as little overlap between the individual leaves as possible to estimate the leaf area using the LeafTraits package in R (Bernhardt-Römermann, unpublished) and dried them to assess specific leaf area (SLA), a proxy for growth rates (Garnier, 1992; Pérez-Harguindeguy et al., 2013). The entire leafy stem was assessed in SLA measurements, as the stems are also photosynthetically active.

2.4. Statistical analysis

To analyse the differences in abiotic factors between natural populations and gardens, we performed Type III ANOVAs to account for the differences in size of the two groups, that is, gardens and natural populations. We used the site conditions LAI, pH, C, N, Mg and P as dependent variables in six separate models and the site (garden or natural population) as an explanatory variable. Here and in all following analyses, we did not distinguish between botanical garden and private garden but grouped them together as "garden" as the same artificial structures were erected in both types of gardens and the management was comparable.

As the locations top and edge of the artificial mounds visually differed, we assessed the differences in temperature above, on and below soil on top of the mounds as well as on the edge and in three natural populations. We performed a Wilcoxon rank sum test with continuity correction. The same was done for soil moisture, as data was not normally distributed.

To test whether the ecophysiological parameters differed between gardens and natural populations, we again performed Type III ANOVAs. For each trait (diameter, F_v/F_m, PI_{abs}, Chl, and SLA) we tested whether the plants differed in between regions (Hrnčíře and Želivka) and gardens (derived from populations from Hrnčíře and Želivka) as well as between the side of the mounds and the top, using them as explanatory variables, resulting in six factor levels. This was done using an ANOVA followed by a TukeyHSD test.

To test, which abiotic factors influenced the performance of the plants in the gardens, we performed linear mixed effect models with garden as random factor and the abiotic site conditions (LAI, pH, C, N, Mg and P), the location on the mound (side, top) as well as the region of origin (Hrnčíře and Želivka) as explanatory variable. All metric variables were scaled prior to the analysis. We also allowed for two-fold interactions between the abiotic site conditions and location on the mound as well as with origin of the population. We then simplified the models *via* backwards selection (Crawley, 2012).

To assess the overall trait syndrome of the populations to see, how they differed, we analysed them in a multivariate trait space and performed a principal component analysis (PCA) with all individuals based on all traits evaluated in the study. Plant traits were standardized prior to analysis. Again, all six types of sites (natural populations in Hrnčíře and Želivka, garden populations derived from Hrnčíře and Želivka split into side and top) were included into the analysis. We also included all abiotic factors measured (LAI, pH, as well as C, N, Mg and *P* values of the soil) *via* the envfit() function. To test their impact, we performed a redundancy analysis including region and type (natural populations *vs.* gardens) as environmental factors (RDA).

All statistical analyses were carried out in R (R Core Team, 2021). Type III ANOVA were performed with the 'car' package (Fox et al., 2012). The PCA and RDA was computed using the 'vegan' package (Oksanen et al., 2007). All graphics were done using 'ggplot2' and 'ggbiplot' for the graphical representation of PCA (Wickham, 2009; Vu, 2011).

3. Results

3.1. Abiotic differences between ex situ conservation sites and natural populations

The abiotic site conditions differed between gardens and natural populations. The LAI was significantly higher in natural forested areas (median ~3.7) than in gardens (0.9; $F_{1,23} = 34.0$; $p < 0.001$; Fig. 2A). The pH value of the soil was significantly lower at natural than in garden sites (pH 5.7 *vs.* 7.0; $F_{1,22} = 21.9$; $p < 0.001$; Fig. 2B) whereas N (1.2 *vs.* 0.3 %), C (22.1 *vs.* 5.4 %) and Mg levels (3635 *vs.* 2093 mg/kg) in the soil were higher in natural populations than in gardens ($F_{1,22} = 56.1$, $p < 0.001$; $F_{1,22} = 52.0$, $p < 0.001$ and $F_{1,22} = 12.1$, $p < 0.001$, respectively; Fig. 2C–E). Soil P content did not significantly differ between gardens and natural populations (Fig. 2F).

We found that temperature on top of the mounds was higher than on the sides (mean temperature 9.7 *vs* 9.3 °C above soil, 9.0 *vs.* 8.6 °C on soil, 8.5 *vs.* 8.3 °C below ground; see Appendix 1). Looking at the average across the measuring time, the gardens were within range of the

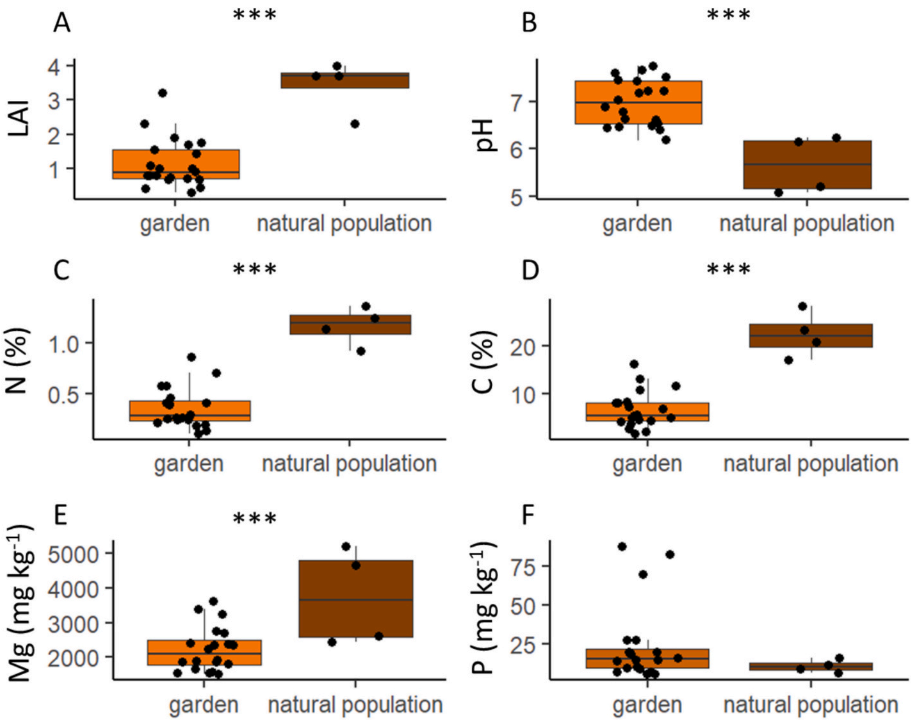

Fig. 2. Comparison between *ex situ* conservation sites and natural populations in A) Leaf area index (LAI), B) pH, C) Soil nitrogen content, D) Soil carbon content, E) Soil magnesium content and F) Soil phosphorous content. Private gardens are displayed in light brown, and natural populations are displayed in dark brown. Asterisks denominate statistical difference between group (***: $p < 0.001$, **: $p < 0.01$, *: $p < 0.05$). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

natural populations below ground, but above ground and on the soil surface, the sides of the mounds were colder than the natural populations. The natural populations within the forests were much warmer in winter whereas in summer they were cooler below soil. In addition to that, soil moisture was extremely different between sites, noticeably in winter and spring where the tops of the mounds were especially dry

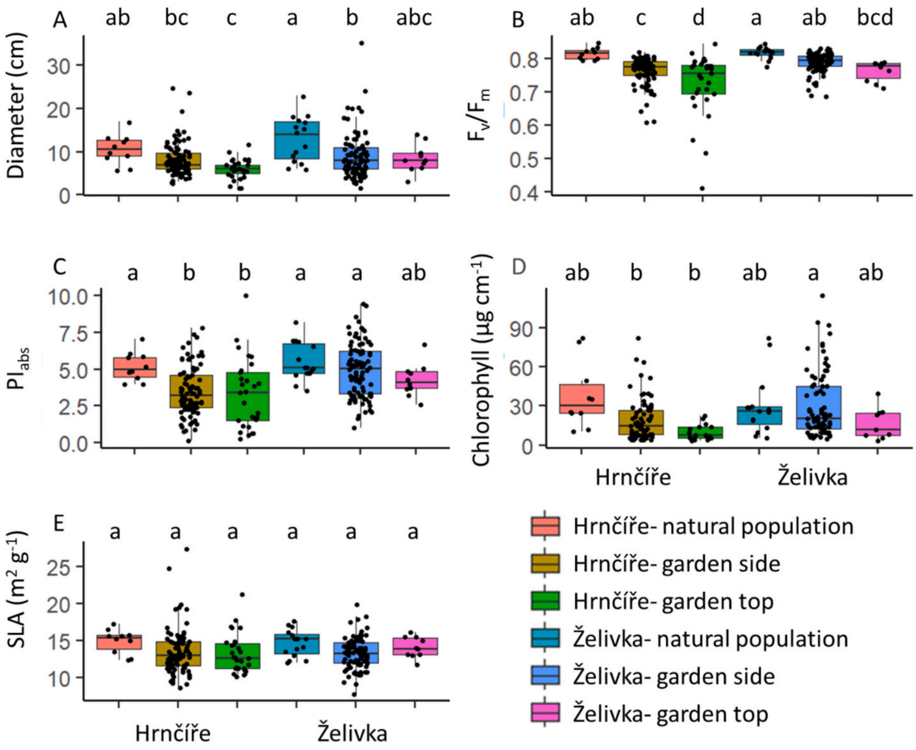

Fig. 3. Comparison between *ex situ* conservation sites and natural populations in A) plant diameter, B) F_v/F_m, C) PI_{abs}, D) chlorophyll content and E) specific leaf area (SLA). Private gardens are displayed in light green, natural populations in dark green. Asterisks denominate statistical difference between group (***: $p < 0.001$, **: $p < 0.01$, *: $p < 0.05$). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

(mean soil moisture on top 1139.8 and on side 1516.6).

3.2. *Plant functional traits between ex situ conservation sites and natural populations*

Plants in natural populations were generally larger than plants in *ex situ* conservation sites (median 11 *vs.* 7 cm), with species grown on the side being slightly larger than the ones on the top. In general, plants from Želivka as well as their offspring were slightly bigger than the plants originating from Hrnčíře, yet this was not significant (overall model $F_{5,\,240} = 5.7$, $p < 0.001$; Fig. 3A). Plants from natural populations were less stressed as can be seen from their higher F_v/F_m (0.82 *vs.* 0.78) with plants from Hrnčíře showing slightly lower values than plants from Želivka. Again, plants grown at the sides of the mounds had slightly higher values than plants grown on top ($F_{5,\,240} = 14.2$, $p < 0.001$; Fig. 3B). Also, PI_{abs} values were higher in natural populations than in the gardens (5.1 *vs.* 3.8, indicating higher performance). However, plants from Želivka and their offspring had slightly higher PI_{abs} values. Plants grown on the side of the mounds and on top did not differ in PI_{abs} ($F_{5,240} = 10.0$, $p < 0.01$; Fig. 3C). This was the same pattern for chlorophyll content (26.3 vs. 16.2 μg cm^{-1}; $F_{5,\,200} = 5.4$, $p < 0.05$; Fig. 3D) whereas there was no difference in SLA between the locations ($p > 0.1$; Fig. 3E).

3.3. *The influence of abiotic conditions on the traits on plant performance*

Within the gardens, we tested the influence of abiotic conditions, the location on the mounds and the origin of populations on plant performance *via* linear mixed effect models. We found that only pH in combination with the origin of the population had an impact on the size of the plants. None of the abiotic factors measured within the gardens had a significant impact on SLA, yet SLA strongly differed between gardens presumably due to abiotic conditions not assessed. PI_{abs} differed between gardens as well but was also affected by the population of origin and its interaction with soil C and P as well as the location on the mound. F_v/F_m hardly differed between gardens but was influenced by soil pH and soil nutrients. Chlorophyll content, however, was only significantly affected by soil P in interaction with population (Table 1).

The natural populations clustered together yet were not very different from the garden populations, which were much more variable (Fig. 4). The first axis, explaining 37.8 % of the variation in the data, was mainly associated with chlorophyll fluorescence, *i.e.*, stress and performance, whereas the second axis, explaining 21.8 % was mainly associated with growth rates, namely SLA. The natural populations showed slightly higher performance (PI_{abs}) and low stress (F_v/F_m) yet did not differentiate in the second axis from the garden plants. Only the abiotic factors LAI, as well as the soil C, N and Mg content proved to be significant in the PCA, all being slightly associated with the first axis.

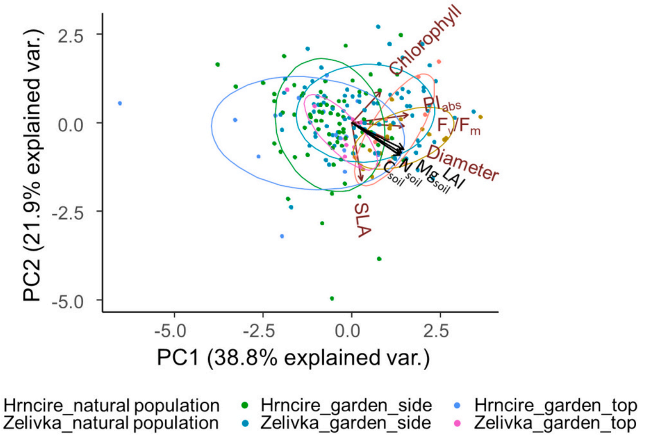

Fig. 4. PCA of all traits based on the location of the species (region, natural population or garden and within garden, top of the mounds or side). The significant environmental parameters are represented ($p < 0.05$). Ellipses represent 95 % confidence intervals.

Table 1

Statistical output of linear mixed effect models using “garden” as random factor. ‘–’ denote factors or interactions, which were not included in the minimum adequate model, ‘n.s.’ factors, which were included but where not significant, ‘.’ Factors which were marginally significant ($p < 0.1$), and asterisks indicate significant levels (‘*’: $p < 0.05$; ‘**’: $p < 0.01$, ‘***’ $p < 0.001$).

	Diameter	SLA	PI_{abs}	F_v/F_m	Chl
LAI	–	–	–	n.s.	n.s.
Soil N	–	–	n.s.	n.s.	n.s.
Soli C	–	–	n.s.	.	n.s.
pH	n.s.	–	–	*	n.s.
Soil Mg	–	–	–	*	n.s.
Soil P	–	–	n.s.	.	n.s.
Location mound	*	–	n.s.	n.s.	n.s.
Regions	.	–	**	n.s.	n.s.
LAI: regions	–	–	–	.	.
pH: regions	*	–	–	–	–
Soil N: location mound	–	–	*	*	–
Soil N: regions	–	–	–	–	n.s.
Soil C: location mound	–	–	*	–	–
Soil C: regions	–	–	**	*	.
Soil P: regions	–	–	*	–	*
Soil Mg: location mound	–	–	–	.	–
Soil Mg: regions	–	–	–	*	n.s.
R^2 marginal	0.09	0.00	0.29	0.32	0.17
R^2 conditional	0.11	0.19	0.47	0.33	0.40

The RDA analysis showed that overall, the included environmental variables explain 11.3 % of the variation in traits. The only significant parameters were region (3 %), soil carbon (8 %) and magnesium (2 %).

4. Discussion

In this study, we presented a novel approach for plant conservation, namely, using private gardens as sites for *ex situ* conservation. Even though the conditions of the sites of the natural populations could not be entirely mimicked *via* the construction of artificial serpentine mounds in the gardens and the plant performance based on plant functional traits was not as good as under natural conditions. 73 % of the plants survived in the first year in the *ex situ* conservation sites, whereas 25 % also survived the following year, but the resulting seeds had a high germination rate (70–95 %, pers. obs.), so many new individuals could establish. We thus created many *ex situ* populations with a good chance of maintaining high levels of genetic diversity providing material for population re-enforcement or reintroductions in the future. Moreover, the differences in abiotic conditions led to a higher variability in plant functional traits than in the natural sites. This is beneficial for the survival of the species, as this variability might be crucial for the establishment in new locations under future conditions.

While at the beginning of the project the total number of individuals of *M. smejkalii* globally was 434, it increased to 1015 individuals in the wild due to management interventions on site such as the revitalisation of the natural habitats and a decrease of vandalism and plant collections due to activities and rising awareness (Pánková, 2021). Additionally, >1100 individuals grow in *ex situ* populations in private gardens and >1500 individuals in reintroduced or re-enforced populations (pers. obs., Siebenkäs et al., in review).

Modelling studies of *M. smejkalii* showed that there is a positive growth rate in some populations and almost no quasiextinction probability suggesting that *M. smejkalii* populations can further expand (Zhu et al., 2021). However, the maximum population spread of <6 cm/year and a maximum seed dispersal of 2.5 m highlights that the spread is very limited and rather slow (Zhu et al., 2021). That in combination with the patchy occurrence of serpentine soils, which is a mandatory precondition for the growth of *M. smejkalii*, makes it difficult for *M. smejkalii* to colonize other potentially suitable sites (Zhu et al., 2021). However, the main species pollinator, hoverflies, can fly for tens of kilometres, so the exchange of pollen is possible. The private gardens could thus provide steppingstones to facilitate migration and pollen dispersal and help to maintain bigger overall populations and also be used for the collection of seeds for reintroduction into the natural habitats and the establishment of new populations. Staude (2024a) demonstrated that dispersal traits between non-native plant species which are known to escape from gardens are not necessarily superior to the ones of endangered native plants and cultivating the natives in private gardens could make the gardens into steppingstones. The likelihood for *M. smejkalii* to establish is however very low as there are hardly suitable serpentine patches in the landscape.

Experiments on *M. smejkalii* have shown that hybridisation with the congener allopatric species *M. caespitosa* is possible and due to their better performance, hybrids might outcompete *M. smejkalii* in the wild (Lozada-Gobilard et al., 2020). Natural hybridization is not possible in nature at the moment since there is no overlap in the species areas – the closest related *Minuartia* species grow in Dokesko (150 km from Želivka) and Krkonoše (120 km apart). However, based on the formerly wider distribution, contact in the past is very likely. The risk of hybridization occurs only in botanical gardens as other *Minuartia* species are often grown there. Since *Minuartia* is not an ornamental plant and is thus not planted in private gardens, the risk is considerably lower there. Additionally, private gardens were checked for the occurrence of other *Minuartia* species prior to planting. This makes private gardens superior as sites of *ex situ* conservation over botanical gardens (Ensslin and Godefroid, 2019; Lozada-Gobilard et al., 2020). Individuals for restoration purposes must thus be selected carefully if plants from *ex situ* conservations are selected to be reintroduced into the wild.

As the species is a serpentinophyte, serpentine mounds have been constructed at each garden and used for planting. These newly constructed mounds were unfortunately not entirely stable due to mechanical perturbance caused by sliding gravel. This led to higher mortality and lower performance of plants on the least stable tops of the mounds. The mounds had lower LAI values, indicating that the gardens were less shaded and the plants more exposed to direct sunlight than the natural populations. This was especially true on top of the hills, where the temperatures were too high and probably water retention too low, which we could also see in soil moisture. Thus, many plants died on top of the hills, whereas they were more vital at the base of the hills where they were a bit more shaded (due to the hill) and more water was available, also due to the vicinity of the surrounding lawns. To mimic natural conditions, gardeners were asked to minimalize the plant watering and only water occasionally, yet as the mounds were more exposed to sunlight than under shaded forest conditions, evaporation was likely higher. Thus, improving water conditions, although the plant is quite drought tolerant, might have been beneficial and the height of the mounds could have been reduced to improve site conditions. Also, additional shading to better capture the conditions within the forests would be a measure to try out to increase the performance of the plants in the gardens. However, the fact that we did not manage to mimic the abiotic conditions in the gardens completely is not necessarily a bad thing, as high variation in abiotic conditions might lead to maintenance of genetic variation in the long run and therefore facilitate the species survival as already indicated by the increased variation in plant functional traits.

In our research we could demonstrate differences between the populations, which illustrates regional differences and adaptations. All *ex situ* populations were established on open sites in the gardens. These conditions simulated, however, only the natural conditions in Želivka, as the forest was much denser in Hrnčíře. Overall, plants from Želivka seemed to perform similarly or better than plants from Hrnčíře. We addressed both remaining populations separately, as *M. smejkalii* was shown to have high genetic diversity within populations, low to moderate differentiation between populations within region and moderate regional differences (Stojanova et al., 2020). Based on genetic analysis and regional differences, the strict separation between the two remaining populations was suggested and was enforced throughout the project (Stojanova et al., 2020). As the species does not show inbreeding depressions, the sites within the natural populations could have a high gene flow (Stojanova et al., 2021). This is important for plant restoration in general, where previous studies pointed out that using regional seeds might be crucial to maintain plant populations adapted to local conditions and keep the genetic diversity within the species high, at least in some species (Hamilton, 2001; Bucharova et al., 2017; Durka et al., 2017; Durka et al., 2024). Concerning these facts, it is on the one hand important to respect the differences between regions and keep plants separated. On the other hand, the gene flow between populations should be supported. Therefore, the seeds from *ex situ* cultures could be used to reinforce small populations. Kottler et al. (2021) even found no adverse effects of even high gene flow from non-local provenances to isolated populations, which would indicate that not necessarily local populations must be used in restauration experiments, so more research is needed there and experiments conducted to test, which the best method for *M. smejkalii* would be.

Using plant functional traits related to ecophysiological performance, we could demonstrate that plants in gardens were less vital than in natural populations. This might be due to the fact that plants in natural population had a long time to adapt to the local conditions, whereas the ones in the gardens were only recently exposed to the novel conditions which deviated from the ones in the natural habitats. Since we evaluated newly established plants though, the differences were caused by environmental conditions rather than by the effects of the transplantation. However, the plants could also have been younger in the gardens and therefore the root system might not have been as developed which might have led to the differences. The plants were not only smaller, which reduces their competitive ability, but were also more stressed, showed lower performance and chlorophyll content and a slower growth rate, which could make them better adapted to natural, stressful environments. This was more pronounced in plants growing on top of the hills, where abiotic conditions were even more different to forest conditions than at the side of the mounds, where temperatures were lower and water availability higher, which further strengthens that explanation.

In a multivariate analysis, we could see that plants from gardens were much more variable than plants in the natural populations and there is an overlap with the individuals collected in the natural populations, thus the different conditions led to a diversification in plant functional traits. Plants from *ex situ* conservation sites often suffer from maladaptions and decreases in plant performance which is due to the low sizes of the population which are maintained or selective sampling of the individuals which are conserved (Raven, 2004; Enßlin et al., 2011; Ensslin et al., 2015; Ensslin and Godefroid, 2019). In our study, plants were planted less than five years ago from seedlings cultivated in plant nurseries, so differences in plant performance are most likely due to differences in abiotic conditions in the gardens. In addition to that, even though the gardens were mostly smaller than the natural populations, there are more sites than in traditional *ex situ* conservation projects, which might mitigate this issue. Also, the number of individuals in many natural sites was smaller than in the gardens.

Even though private gardens may sometimes inadvertently harbour endangered or declining plant species, there are few programs to cultivate endangered or threatened endemic plants in private gardens. One exception is the ‘Topos’ project, where plants are grown in private gardens for seed production to reintroduce them into the wild in Switzerland (www.toposmm.ch). There are also initiatives for studying home gardens for (agricultural) biodiversity maintenance with a special focus on food security (Galluzzi et al., 2010; Idohou et al., 2014, Korpelainen, 2023). Additionally, urban conservation gardening has been proposed to cultivate native and endangered plant species in urban greenspaces and private gardens (Segar et al., 2022; Munschek et al.,

2023) and also initiatives to get commercial seed producers to offer native plant species have been launched as listed in Munschek et al. (2023). Current research shows that both, botanical gardens and private gardens, are important sites for *ex situ* plant conservation (Ismail et al., 2021). Often, gardens are considered as too small for conservation actions, yet they are a lot and depending on the size and needs of the species, the limited space available might be plenty as in the case of *M. smejkalii*, where the smallest mounts were $<2\ \mathrm{m}^2$. Even traffic islands and urban green spaces can be used as potential conservation sites (Ismail et al., 2021) as demonstrated for example in Wellington, New Zealand (Sawyer, 2005). These sites automatically create resources for research and education.

Private gardens also offer the advantage that the plants are mostly protected from browsing and grazing animals (Sawyer, 2005), which might be an important detrimental factor in the wild yet might be more susceptible to slugs which can cause detrimental damage. A close partnership between the horticultural industry, plant collectors, and private gardeners to determine optimal growth conditions of plants and engage the local public can help to use the full potential to establish private gardens as *ex situ* conservation sites and to ensure plant survival in times of accelerated species extinction (Ismail et al., 2021; Segar et al., 2022; Munschek et al., 2023).

Of course, using private gardens instead of botanical gardens comes with different pitfalls, as they are rarely run by professional gardeners and the care is thus based only on their own expertise, capacity and voluntariness. Additionally, private gardens might change hands more often which comes with different ideas on garden management; Whereas, in botanical gardens the cultivation continues after the change of curator, the cultivation in local gardens is depending on the owner. If the gardeners die or the garden is sold, it could lead to abandonment of the *ex situ* cultivation. Therefore, it must be treated very carefully in the contract prior to the establishment. In our experience, only keen gardeners volunteered in the first place, yet it is important to set up a dynamic network which helps to include new gardens any time to allow for potential losses of *ex situ* conservation sites while still enduring a sustainable development of the populations and adding to the variability of the populations as each new garden comes with slightly different abiotic conditions.

All activities, including information campaigns on the species such as postcards, DIY jewels, environmental education programs as well as media outputs which were distributed and promoted in this project presented here in addition to the *ex situ* conservation led to higher awareness in the local communities and also decreased vandalism (Pánková, 2021). Thus overall, we achieved an amelioration of the conditions for *M. smejkalii* and created effective and self-sustainable *ex situ* conservation sites where we can also collect sufficient plant material for species reintroduction into wild stands (pers. obs.)

The study demonstrates that ‘Rescue planting in private gardens’ is suitable for species conservation. Conserving species with less specific habitat demands might have been easier, yet many endangered species do have very restricted optimal growing conditions. However, with this study, we could show that *ex situ* conservation in private gardens increased the source of seeds, which can be used for species reintroduction or population reinforcement. Additionally, it increased the awareness of the local people. Currently, subsequent projects for using these seeds for establishment of new populations in the field are ongoing. Cultivation of plants in private gardens has also been approved as a possible methodology for species conservation in rescue programs in the Czech Republic and can thus be used for other species in the future. At the same time, the populations maintained high variation suggesting that this approach might be a valuable addition to conservation in a single site in botanical gardens.

CRediT authorship contribution statement

Solveig Franziska Bucher: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Zuzana Münzbergová:** Writing – review & editing, Resources, Methodology, Funding acquisition, Data curation, Conceptualization. **Hana Pánková:** Writing – review & editing, Project administration, Methodology, Funding acquisition, Data curation, Conceptualization.

Declaration of competing interest

None to declare.

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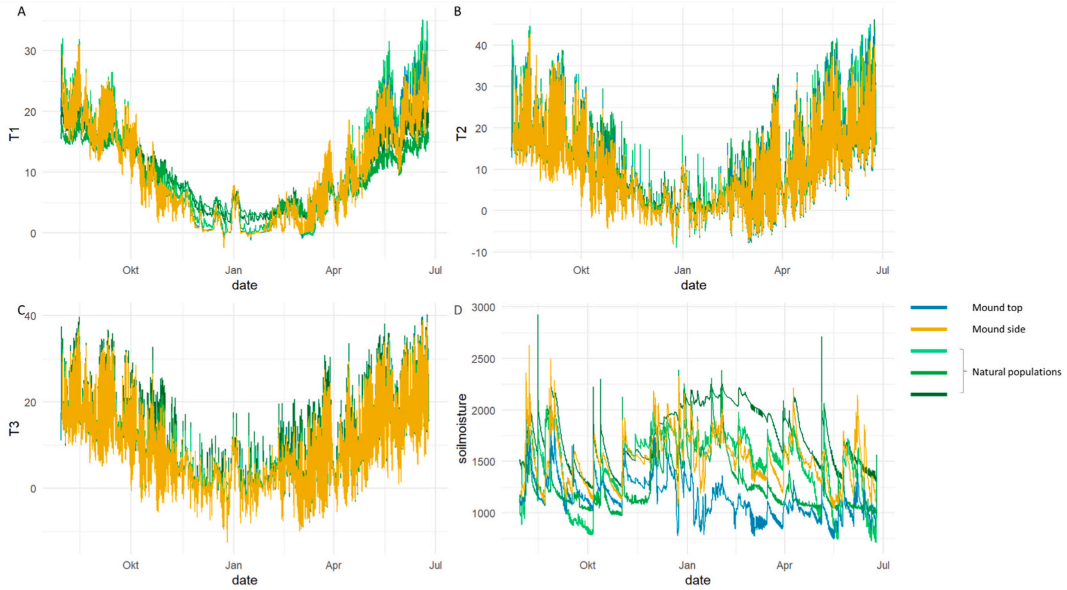
Appendix 1. **A) Temperature data 6 cm below soil, B) Temperature 2 cm above soil, C) Temperature 15 cm above soil, D) soil moisture of the time period July 31, 2021–June 23, 2022. Data on top of the mounds is displayed in turquoise, at the side of the mound in yellow. The three natural sites are displayed in green. The darker the green, the more shaded the area (light green DK2_SK9, mid-green DK5, dark green DK5)**



Data availability

Data will be made available on request.

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