



How to not trade water for carbon with tree planting in water-limited temperate biomes?



Csaba Tölgyesi^{a,b,*}, Alida Anna Hábenczyus^a, András Kelemen^{a,c}, Péter Török^{d,e}, Orsolya Valkó^c, Balázs Deák^c, László Erdős^{d,f}, Benedek Tóth^a, Nándor Csikós^{b,g}, Zoltán Bátori^a

^a Department of Ecology, University of Szeged, Közép fasor 52, Szeged 6726, Hungary

^b MTA-SZTE Lendület Applied Ecology Research Group, Közép fasor 52, Szeged 6726, Hungary

^c ÖK Lendület Seed Ecology Research Group, Institute of Ecology and Botany, Centre for Ecological Research, Alkotmány utca 2-4, Vácrátót 2163, Hungary

^d ELKH-DE Functional and Restoration Ecology Research Group, University of Debrecen, Egyetem tér 1, 4032 Debrecen, Hungary

^e Polish Academy of Sciences, Botanical Garden - Center for Biological Diversity Conservation in Powśin, Prawdziwka St., 202-973 Warszawa, Poland

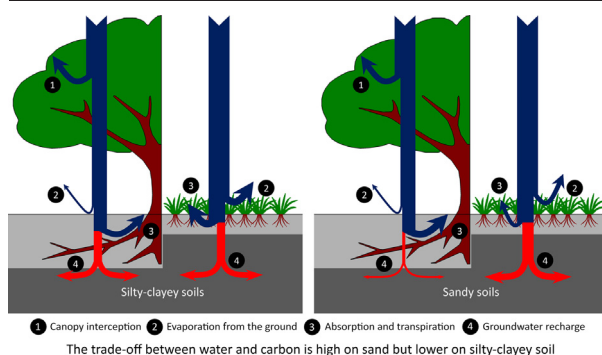
^f Institute of Ecology and Botany, Centre for Ecological Research, Alkotmány utca 2-4, 2163 Vácrátót, Hungary

^g Department of Soil Mapping and Environmental Informatics, Institute for Soil Sciences, Centre for Agricultural Research, Herman Ottó út 15, Budapest 122, Hungary

HIGHLIGHTS

- Tree planting is a popular nature-based solution for climate change mitigation.
- Afforestation in water-limited ecosystems aggravates water scarcity.
- We show that the strength of the carbon vs. water trade-off depends on soil texture.
- Water loss is much higher on sandy soils than on fine-textured soils.
- Tree planting efforts should avoid sandy regions and focus on fine-textured soils.

GRAPHICAL ABSTRACT



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ABSTRACT

The most widespread nature-based solution for mitigating climate change is tree planting. When realized as forest restoration in historically forested biomes, it can efficiently contribute to the sequestration of atmospheric carbon and can also entail significant biodiversity and ecosystem service benefits. Conversely, tree planting in naturally open biomes can have adverse effects, of which water shortage due to increased evapotranspiration is among the most alarming ones. Here we assessed how soil texture affects the strength of the trade-off between tree cover and water balance in the forest-steppe biome, where the global pressure for afforestation is threatening with increasing tree cover above historical levels. Here we monitored vertical soil moisture dynamics in four stands in each of the most common forest types of lowland Hungary on well-drained, sandy (natural poplar groves, and *Robinia* and pine plantations) and on poorly drained, silty-clayey soils (natural oak stands and *Robinia* plantations), and neighboring grasslands. We found that forests on sand retain moisture in the topsoil (approx. 20 cm) throughout the year, but a thick dry layer develops below that during the vegetation period, significantly impeding groundwater recharge. Neighboring sandy grasslands showed an opposite pattern, with often dry topsoil but intact moisture reserves below, allowing deep percolation. In contrast, forests on silty-clayey soils did not desiccate lower soil layers compared neighboring grasslands, which in turn showed moisture patterns similar to sandy grasslands. We conclude that, in water-limited temperate biomes where landscape-wide water regime depends on deep percolation, soil texture should drive the spatial allocation of tree-based climate mitigation efforts. On sand, the establishment of new forests should be kept to a minimum and

* Corresponding author at: Department of Ecology, University of Szeged, Közép fasor 52, Szeged 6726, Hungary.
E-mail address: festuca7@yahoo.com (C. Tölgyesi).

grassland restoration should be preferred. The trade-off between water and carbon is less pronounced on silty-clayey soils, making forest patches and wooded rangelands viable targets for both climate mitigation and ecosystem restoration.

1. Introduction

One of the most widespread and popular nature-based solutions for mitigating anthropogenic climate change is tree planting (Bond et al., 2019; Holl and Brancalion, 2020; Seddon et al., 2021; Tölgyesi et al., 2022). It is based on the assumption that trees, if planted in large quantities globally, can decisively counteract anthropogenic CO₂ emissions by capturing and storing carbon (Bastin et al., 2019). This new hope stimulated various platforms, including intergovernmental, governmental and non-governmental ones, to adopt tree planting as their climate action (Brancalion and Holl, 2020; Selva et al., 2020). Tree planting is often called forest restoration, and the technological and terminological distinction from afforestation is blurred, although it is a key difference to guide us in planning activities (Lewis et al., 2019). Forest restoration is a type of ecosystem restoration, where the target is a natural forest type that has been degraded or totally cleared from a specific location (cf. van Andel and Aronson, 2012). Forest restoration when meant this way improves ecosystem services for humans and ecosystem conditions for nature, while adverse side-effects are rarely reported (Di Sacco et al., 2021).

In contrast with forest restoration, afforestation does not assume that the planted forest will approach the natural type of forest, thus the target forest can also be a non-native plantation forest, and it is not necessarily assumed that the environmental conditions support the occurrence of natural woody vegetation and whether there used to be a forest in the that specific region whatsoever. Tree planting actions, however, target such historically open landscapes too, that is, actions do not necessarily aim to restore natural ecosystems, and may even transform natural open ecosystems, such as ancient grasslands and savannas (Veldman et al., 2015; Bond et al., 2019). Such land cover transformations have far reaching adverse effects including altered fire regime, albedo, soil salinity, changes in native biodiversity (Temperton et al., 2019; Veldman et al., 2019; Fleischman et al., 2020; Mujica et al., 2021) and, probably most importantly, water regime (Jackson et al., 2005; Li et al., 2021; Liu et al., 2022). But are these acceptable costs to mitigate climate change? Advocates of open ecosystems claim that they are not, and afforestation of these ecosystems is not even necessary to lock carbon away from the atmosphere, as open ecosystems often store large quantities in the soil in a safe, fire-resistant form, which would actually be compromised by tree planting (Dass et al., 2018; Terrer et al., 2021).

In large biogeographical regions of the world, such as the temperate forest-grassland mosaic ecosystems of Eurasia (i.e., the forest-steppe; Erdős et al., 2018; Feurdean et al., 2018), North-America (prairie-forest ecotones and ‘parkland’ ecosystems; Barbour and Billings, 2000; Teed et al., 2009) and also in South-America (pampa-forest ecotones; Mancini, 2009) stands of both forest and grassland can be self-sustaining, natural habitats (Pausas and Bond, 2020). That is, both ecosystem states may be targets of ecosystem restoration, which may represent a dilemma and raise potential conflicts between pro-afforestation groups and their opponents.

Large tracts of natural forest-grassland mosaic ecosystems in temperate biomes have historically been cleared and transformed into arable fields (Gowda et al., 2012). Recent trends, however, indicate that many of these croplands are losing their economic profitability, leading to high rates of abandonment (Hobbs and Cramer, 2007; Schierhorn et al., 2013). This, combined with the increasing pressure of tree planting, and the global demand for ecosystem restoration (see e.g. the UN Decade on Ecosystem Restoration announced for the 2020s; Fischer et al., 2021), can threaten with a bias for the forest ecosystem state as the target of ecosystem restoration,

making the activities more afforestation than forest restoration, with all its potential adverse effects.

One of the main reasons why these regions had not fully been forested in historical times is climatic water deficit (Erdős et al., 2018). This limitation continues to persist until present times, and is even exacerbated by ongoing climatic change (Mátyás and Sun, 2014). There is a growing literature that increasing the forest cover can also aggravate water scarcity due to higher evapotranspiration rates compared to open sites in these regions (Farley et al., 2005; Li et al., 2021). Putting it simply, land managers may need to choose between carbon and water when planning to change the current land cover type to forested vegetation (Jackson et al., 2005), and, in line with this trade-off, regions where the highest water costs are associated with tree planting should be avoided in favor of those where the loss of water is lower relative to the carbon gain (Tölgyesi et al., 2021).

The water balance – land cover relationship in water-limited regions is affected by various parameters, such as the annual distribution of the precipitation, slope inclination, and soil depth (Wilcox et al., 2017; Jin et al., 2020; Knighton et al., 2020). Soil texture is another physical parameter that may have an important interactive effect via modifying infiltration rate and water storage capacity (Ma et al., 2016; Marquart et al., 2020), although it is unclear how it affects the optimal forest-grassland balance in restoration targets. The aim of this study is to fill this knowledge gap in the spatial allocation of tree-based climate mitigation programs in water-limited temperate regions worldwide.

Specifically, we had the following three aims:

- (i) Disentangling the hydrologic effects of adjacent forest and grassland habitats in the lowland forest-steppe ecoregion of Hungary, East-Central Europe on soils with fine (i.e. poorly drained, mostly silty-clayey soils) and coarse (i.e. well-drained, mostly sandy soils) texture. For this, we monitored soil moisture relations in the unsaturated zone of the soil, and assessed contributors to water loss into the atmosphere using a comprehensive set of land-based proxies, including tree canopy cover, herb layer biomass, litter density and microclimatic conditions.
- (ii) Using the results to understand the effects of forest and grassland habitats on groundwater recharge and, through this, providing improved guidelines for the dilemma of forest or grassland restoration.
- (iii) Examining the feasibility of our recommendations in light of the current forest cover proportions of water-limited temperate regions.

2. Material and methods

2.1. Study area

The study was carried out in the Great Hungarian Plain of East-Central Europe, composed of a flat lowland region with a size of approx. 52,000 km², most of which is located in Hungary. The climate is subcontinental with Mediterranean influence; mean annual precipitation and temperature are 500–550 mm and 10–11.5 °C, respectively. Winters are moderately cold, summers are hot, and most of the precipitation occurs in early summer, during the growing season (Dövényi, 2012; Tölgyesi et al., 2016). Substrates are either sandy deposits or finer grained, mostly silty-clayey aeolian or alluvial deposits. Sandy soil (mostly Arenosols) is most prevalent in two compact alluvial fans, the Kiskunság Sand Ridge and the Nyírség Sand Ridge, while silty-clayey soils (mostly Phaeozem and Chernozem) make up the rest of the region (Várallyay, 1985). Climate in most of the region does not allow the growth of closed-canopy forests, which is

confirmed by the presence of species of open ecosystems throughout the Holocene (Feurdean et al., 2018) and the abundance of endemic grassland specialist plants, such as *Centaurea scabiosa* subsp. *sadleriana* (Janka) Asch. & Graebn. and *Dianthus serotinus* Waldst. & Kit. The proportion of forest and grassland is a question of debate, but it definitely varied according to moister and drier periods and millennia of fluctuating human influence as corroborated by palynological evidence (Magyari et al., 2010). Presently, most of the Plain is agricultural land interspersed with semi-natural grasslands and forest plantations, which are often composed of non-native trees. Semi-natural, native forests are extremely scarce.

We selected four localities in the eastern part of the Plain with silty-clayey soil, where mature forests of the commonly planted tree species as well as semi-natural grasslands occur right next to each other on identical micro-topographic elevations to make vegetation cover the only apparent difference among the sites within a locality (Fig. 1). Forests were represented by the most common native species, pedunculate oak (*Quercus robur*) and the most commonly planted, non-native species, *Robinia pseudoacacia*. Grasslands were tall swards dominated by *Alopecurus pratensis*, *Festuca pseudovina* and *Poa angustifolia*. Four localities were also selected in the Kiskunság Sand Ridge using the same principles as for silty-clayey soils. Studied forest types included all major tree species of the region: non-native pine plantations (*Pinus nigra* and *P. sylvestris*), native poplar forests (*Populus alba*), and non-native *Robinia pseudoacacia* plantations. The grassland habitat next to the sandy forests was semi-natural bunchgrass steppe, dominated by *Festuca vaginata* and *Stipa capillata*. The tree species represent a gradient of annual canopy lifetime, as pine is evergreen, poplar and oak flush leaves early in the spring, while *Robinia* does so only later in the spring and sheds leaves earlier at the end of the season than poplar and oak. Pine is not economically profitable on silty-clayey soil in the region; therefore, it is not planted. The average distances between the four localities on silty-clayey and sandy soils were 17.8 ± 6.3 km (mean \pm SD) and 16.6 ± 8.0 km, respectively.

2.2. Data collection

Soil texture was assessed within each study site by collecting two soil samples from the topsoil (0–10 cm, below the particulate litter layer) and from a deeper (60–70 cm) layer in 2021. Grain size distribution was measured in an accredited soil laboratory. The particle size ranges we used for sand, silt and clay classification were >0.05 , 0.05–0.002 and <0.002 mm, respectively.

We drilled two 1 m deep holes in each stand of the studied habitats four times over a year in 2018 (sandy sites) and 2020 (silty-clayey sites), and measured volumetric moisture content in three replicates at every 10 cm

depth layer as we proceeded downwards. Measurement times followed the annual cycle of the vegetation, that is, there was one in March (late winter), May (late spring), August (late summer) and November (mid-autumn). Holes within the same forest or grassland stand were at least 20 m apart, so different tree individuals determined their moisture regime. We used a Fieldscout TDR 300 soil moisture meter for the measurements. We considered the upper 100 cm of the soil enough to adequately represent infiltration dynamics as all studied tree species and their relatives in similar water-limited ecosystems have the highest root density, including fine roots, at soil depths within 20 and 80 cm (Hoffmann and Usoltsev, 2001; Cao et al., 2007; Klein et al., 2014). Measurements were done in periods with no significant precipitation during the preceding week.

The actual evapotranspiration capacity of trees depends on the phenological phase of their canopy. We estimated this feature by taking canopy photos with a digital camera in at least two localities of every forest stand from 50 cm above the soil surface at the same times as the soil moisture measurements were done, and estimated canopy cover from them by turning the photos into black and white with manual thresholding using GNU Image Manipulation Software (GIMP 2.10.14). Black pixels corresponded to tree parts (leaves, branches and trunks), and white pixels to the sky. We used the proportion of black pixels in the images as an estimate of canopy cover. Although non-photosynthesizing organs do not contribute to transpiration, they can have an important role in intercepting precipitation and, hence, in evaporation. We assumed 0% canopy cover in the grasslands and did not take photos in them.

We measured herb layer green biomass and litter layer density and used them as a proxy for the evapotranspiration related to the understory of forests and the vegetation of the grasslands. Specifically, we collected all living and dead biomass (separated later in the lab) in four randomly distributed 50 cm \times 50 cm quadrats in every study site in the summer of 2021, and, after drying the samples for four days in a drying chamber at 40 °C, measured their weight with a precision of 0.1 g.

The last parameter we used to interpret water dynamics was microclimate. We assumed higher potential evaporation in warmer microclimate than under cooler conditions. We used Voltcraft DL-121TH data loggers (precision: 0.1 °C) installed 2–3 cm above the ground and covered them by plastic roofs to avoid direct insolation. We installed one sensor to each site at the four sampling dates and recorded microclimate for a full 24-h period with mostly bright sky and little wind.

2.3. Data analysis

We prepared linear mixed-effects models for soil moisture, canopy cover, green biomass and litter. For soil moisture, records were lumped

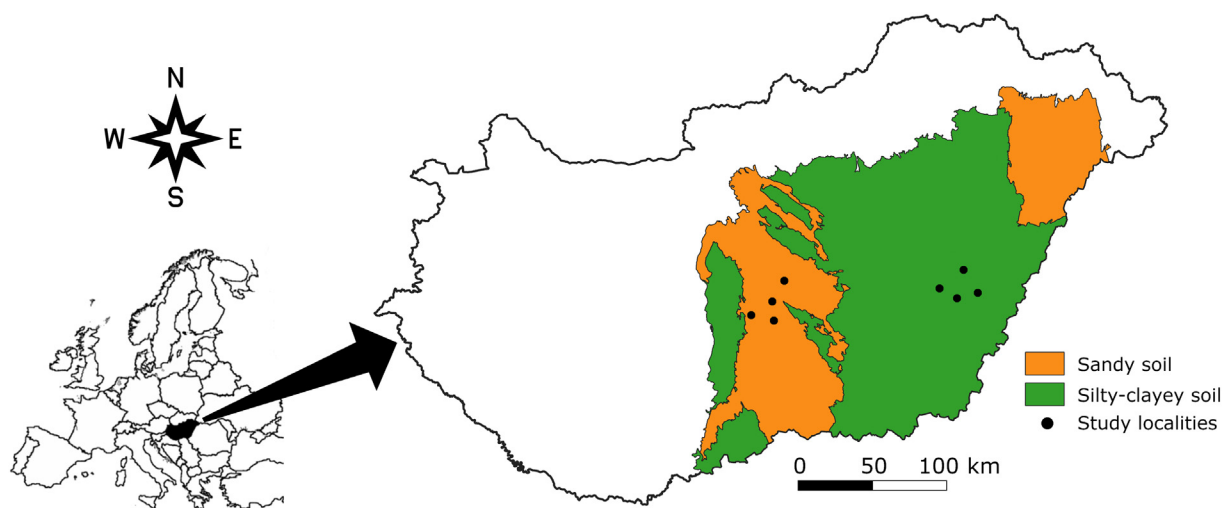


Fig. 1. Location of the study sites in the Kiskunság Sand Ridge and on the silty-clayey region of eastern Hungary. Soil type is shown only in the water-limited plains of the country, where the potential vegetation is a mixture of forested and grassy communities, called the forest-steppe.

into 20 cm thick layers (6 records for each layer of each hole). Regions and seasons were treated separately for all variables, that is, we prepared a total of eight models for canopy cover, (2 regions \times 4 seasons), 40 models for soil moisture (2 regions \times 4 seasons \times 5 layers) and two models for biomass and litter (2 regions). Locality was used as a random effect for canopy cover, biomass and litter, while, we applied hole nested in locality as the random effect for soil moisture to account for the multiple records within each layer of each hole. For temperature records, we used only visual representation.

We performed all analysis in R 3.6.1 statistical environment. We used the 'lmer' function of the lme4 package (Bates et al., 2015) to prepare the models, the 'Anova' functions of the car package (Fox and Weisberg, 2019) to test model significance and the 'emmeans' function of the emmeans package (Lenth, 2021) to calculate pairwise comparisons with adjusted p -values following the Tukey's method. For the visualization of the soil texture on soil triangles, we used the 'TT.plot' function of the soiltexture package (Moeys, 2018).

3. Results

3.1. Soil texture and moisture

Soil texture analysis from the topsoil (0–10 cm) and the lower layers (60–70 cm) confirmed the preliminary categorization of soil of the sites into fine-grained (mostly silty-clayey) and coarse-grained (sandy), and indicated that there is no fundamental difference between the top and lower soil layers of any study site and among the habitat types of each region. Oak forests had somewhat higher clay content particularly in the topsoil, while two *Robinia* stands were a bit loamier in the deeper layer (Fig. 2).

In the silty-clayey sites, the topsoil (0–20 cm) moisture content did not differ among the habitats (i.e. grassland, oak forest and *Robinia* forest) after the winter months, but grasslands were moister than oak forests below this layer (Table S1, Fig. 3). *Robinia* forests turned also moister than oak forests but only below 40 cm. Grasslands and *Robinia* forests did not differ in soil moisture in any depth.

Grasslands lost a notable amount of moisture during the spring months and became drier than oak forests in the upper 40 cm of the soil, and retained higher moisture content than oak forests only at 80–100 cm. *Robinia* forests also turned drier and had similar levels as grasslands in the upper 40 cm, while below this layer grasslands remained moister. *Robinia* forests appeared drier than oak forests in the upper 60 cm and they did not differ below this depth.

Summer rains improved the moisture content of the grasslands, making them moister than oak forests below 40 cm deep, but at the topsoil, oak forests remained moister. *Robinia* forests did not differ from grasslands in the topsoil but below that, grasslands were moister. Oak forests were moister than *Robinia* forests in the upper 40 cm but did not differ beneath. The soil of the habitats recharged during the autumn months, and we found no significant difference between any habitat pair in any depth layer.

In the sandy sites, grasslands were drier in the topsoil (0–20 cm) than poplar and *Robinia* forests after the winter months, but below this, they had similar moisture content (Table S2, Fig. 3). In contrast, pine forests did not differ from grasslands in the topsoil, and appeared drier at 20–60 cm. Pine forests were drier than poplar forests in the upper 60 cm. Pine forests tended to be also drier than *Robinia* forests but the difference was statistically different only at 40–60 cm.

At the end of spring, the topsoil in the grasslands was still drier than in poplar and *Robinia* forests but below 20 cm deep, both forests were drier. Pine forests did not differ from grasslands in the topsoil but below that, pine forests were much drier. Pine forests were also drier than poplar forests in the upper 80 cm and we could confirm significantly lower soil moisture than in *Robinia* forests at depths of 0–20 cm and 40–80 cm. *Robinia* and poplar forests did not differ in any depth layer.

At the end of summer, the topsoil in the grasslands was drier than in the *Robinia* forests but did not differ from poplar and pine forests. Below this, i.e. 20–100 cm deep, all forests were drier than grasslands. *Robinia* forests remained moister than pine forests in the upper 60 cm, while poplar forests were moister than pine only in the topsoil. The statistics confirmed significant difference between poplar and *Robinia* at 20–40 cm deep, with *Robinia* being the moister.

Autumn months allowed some moisture recharging in the forest soils. All forests had moister topsoil than grasslands. Pine still had lower moisture content 40–100 cm deep than grasslands but poplar and *Robinia* caught up on grasslands in all layers; the only difference we detected was that *Robinia* was still drier at 80–100 cm deep than grasslands. Pine was drier than poplar below 40 cm, while the statistics confirmed drier conditions in pine than in *Robinia* only at 60–80 cm deep. Poplar and *Robinia* did not differ significantly in any soil layer.

3.2. Canopy cover

In the silty-clayey sites the canopy cover of the forest types did not differ at the end of the winter ($Chi^2 = 0.13$, $p = 0.723$) (Fig. 4). By the end of spring oak forests flushed their leaves but *Robinia* started it much later, leading to significantly lower scores ($Chi^2 = 69.30$, $p < 0,001$). By the

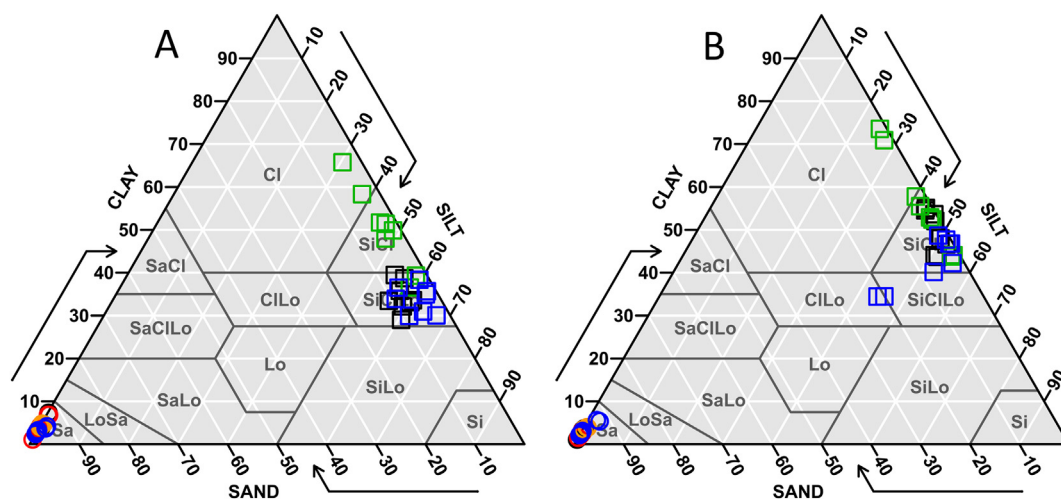


Fig. 2. Soil texture of the top (0–10 cm; A) and lower (60–70 cm; B) soil layers of the study sites. Circles: Kiskunság Sandy Ridge, squares: Eastern Great Hungarian Plain. Black: grassland, golden: pine (mostly overplotted though), red: poplar, blue: *Robinia*, green: oak, Sa: sand, Lo: loam, Si: silt, Cl: clay. Background categories follow the soil taxonomy of the United States Department of Agriculture (Moeys, 2018).

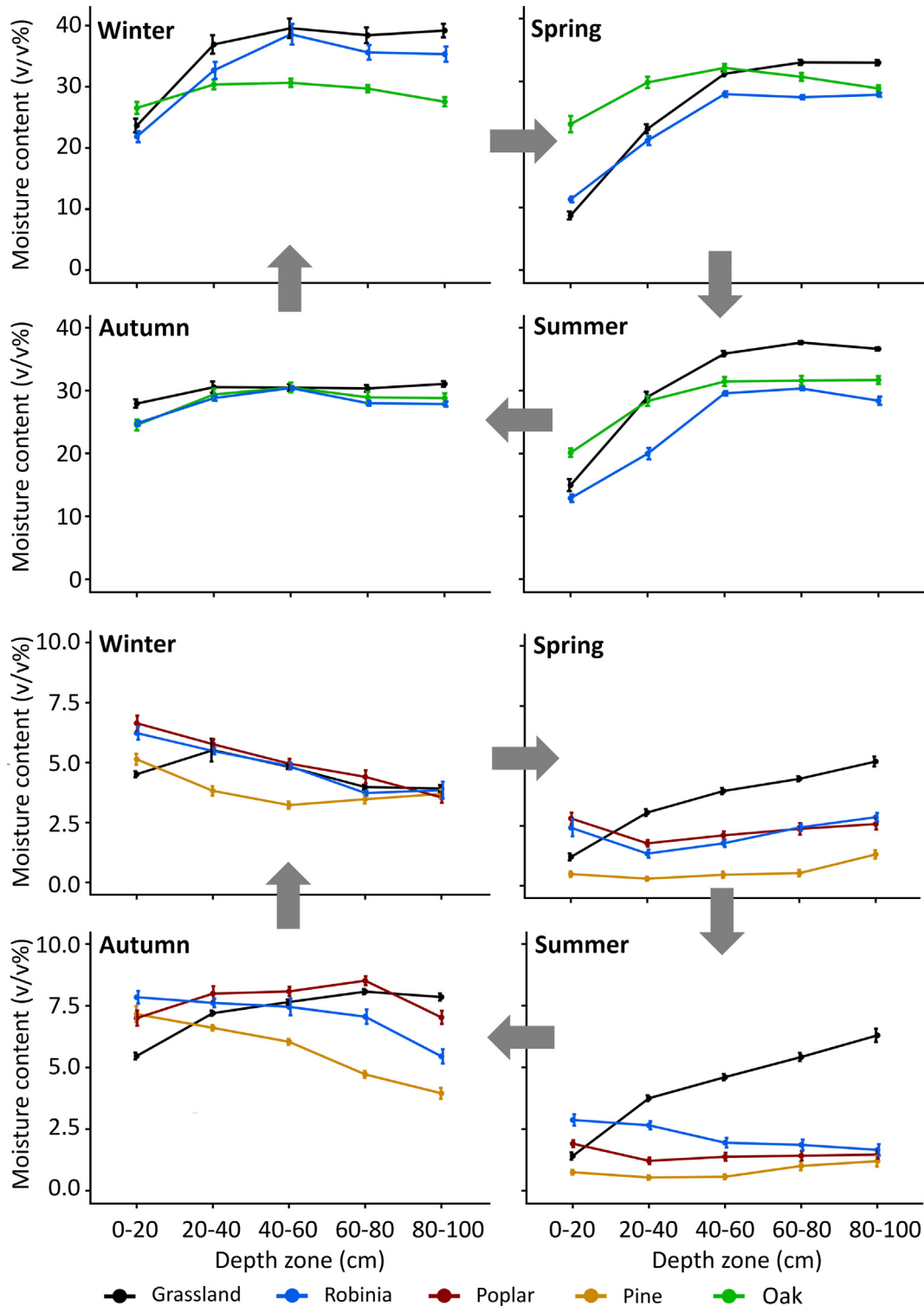


Fig. 3. Soil moisture content of the silty-clayey sites after the winter, spring, summer and autumn periods and the sandy sites after the winter, spring, summer and autumn periods. Whiskers indicate standard errors of the means. For significant differences see text and Table S1.

end of summer, *Robinia* had finally caught up on oak, and the difference disappeared ($Chi^2 = 0.24, p = 0.621$), but later in the autumn *Robinia* shed leaves faster than oak ($Chi^2 = 40.79, p < 0,001$).

In the sandy sites, all models found significant differences among the canopy cover scores of the studied forest types in every season (Table 1, Fig. 4). At the end of winter, pine, being an evergreen tree, had higher cover than *Robinia* and poplar. Poplar had higher scores than *Robinia*, mostly due to the catkins that come out well before the leaves. Unlike in

silty-clayey sites, the canopy cover of *Robinia* had been able to reach its full potential by the end of spring, and its cover even exceeded pine, although by a few percent only on average. At the end of the summer, all forests had high canopy cover values, although the statistics confirmed significant differences among them with *Robinia*, poplar and pine in this decreasing order, but the actual differences seemed negligible in an ecological sense. In autumn, *Robinia* shed its leaves faster than poplar, while the canopy cover of pine remained constant.

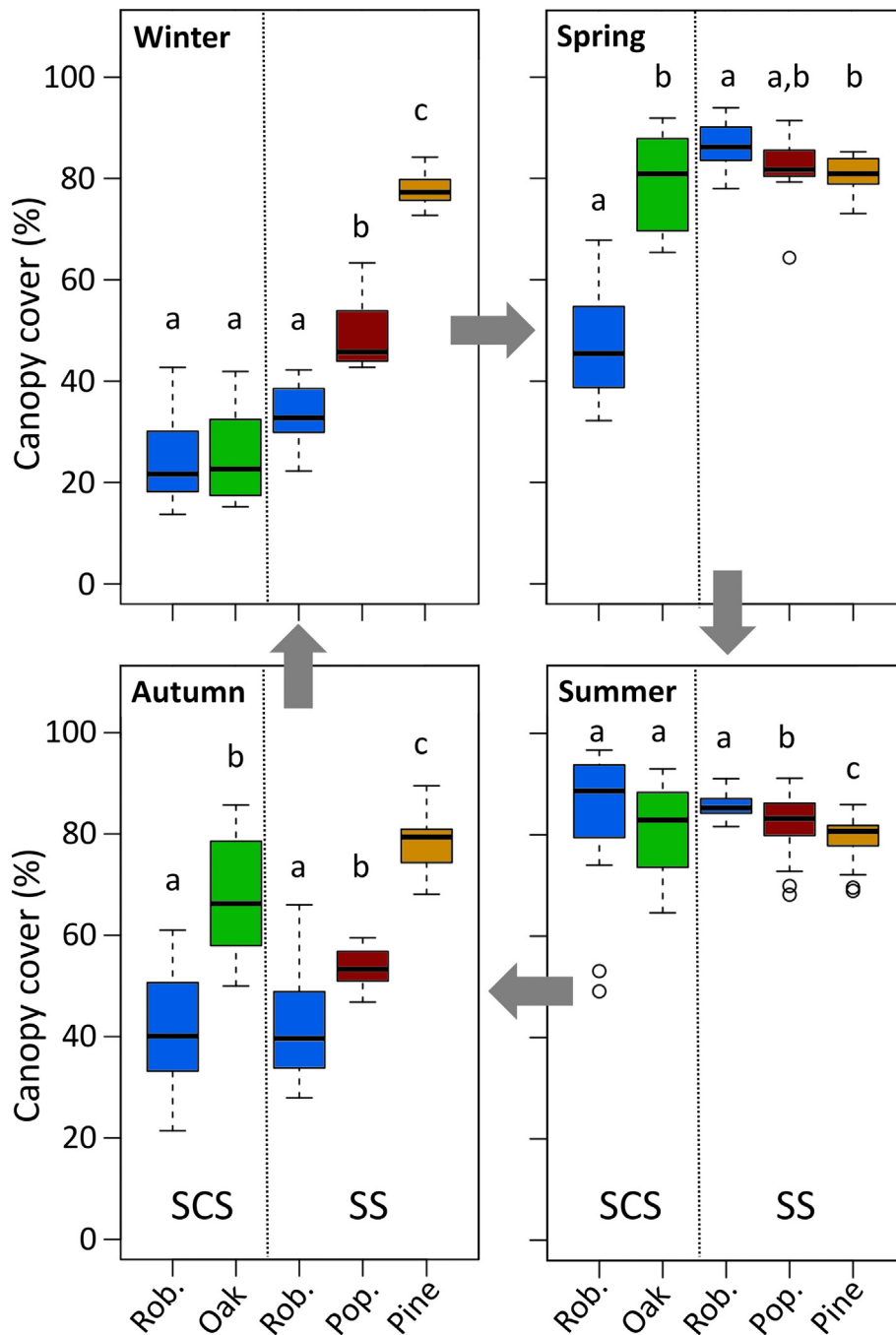


Fig. 4. Canopy cover of forest types at the end of winter, spring, summer and autumn. SCS: silty-clayey sites, SS: sandy sites. Significantly different groups within soil types of each season are marked with different lower case letters. Boxes are interquartile regions (IQR), whiskers indicate the lower and upper quartiles, but are not longer than 1.5-times the IQR; data beyond these are shown as outliers (empty circles). Rob.: *Robinia* forest, Oak: oak forest, Pop.: poplar forest, Pine: pine forest. Tree canopy is absent in grasslands; therefore, they are not shown in the figure.

Table 1

Test results of the linear mixed-effects models of canopy cover scores on sandy soil. Significant results ($p < 0.05$) are marked with boldface. Pine: pine forest, Pop: poplar forest, Rob: *Robinia* forest.

	Winter		Spring		Summer		Autumn	
	<i>Chi</i> ²	<i>p</i>	<i>Chi</i> ²	<i>p</i>	<i>Chi</i> ²	<i>p</i>	<i>Chi</i> ²	<i>p</i>
Model	218.92	<0.001	8.14	0.017	30.28	<0.001	283.28	<0.001
	<i>t ratio</i>	<i>p</i>	<i>t ratio</i>	<i>p</i>	<i>t ratio</i>	<i>p</i>	<i>t ratio</i>	<i>p</i>
Pine - Pop	9.41	<0.001	-0.42	0.906	-2.44	0.045	11.25	<0.001
Pine - Rob	14.59	<0.001	-2.60	0.036	-5.49	<0.001	16.47	<0.001
Pop - Rob	5.19	<0.001	-2.23	0.082	-3.05	0.009	5.21	<0.001

3.3. Herb layer biomass and litter density

In the silty-clayey sites, herb layer biomass at the peak of the growing season (June) was highest in the grassland, but *Robinia* was not far behind it; the herb layer was very scarce in the oak forests (Fig. 5). In general, sandy sites had much lower herbaceous biomass than the silty-clayey sites, and *Robinia* had higher values than grasslands. Both poplar and pine forests had very scarce herb layer. Litter generally followed an opposite pattern, as on silty-clayey soil, the highest amount was found in the oak forests, followed by *Robinia* and ending with grasslands. On sand, pine had the highest amount of litter, followed by poplar and then *Robinia*. Sandy grasslands, similar to the silty-clayey ones had barely any litter (see Table 2).

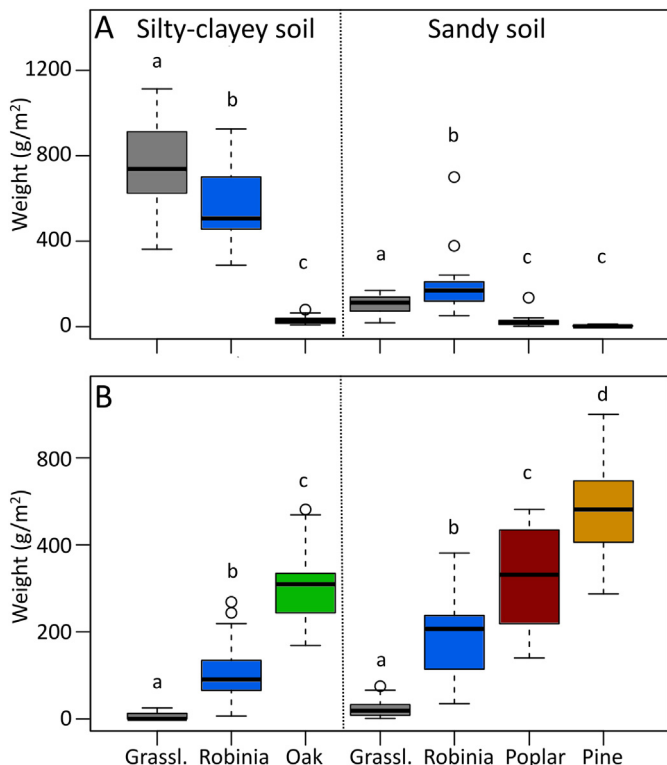


Fig. 5. Herb layer biomass (A) and litter amount (B) of the studied habitats in June, at the peak of the growing season. Significantly different groups within soil types are marked with different lower case letters. Boxes are interquartile regions (IQR), whiskers indicate the lower and upper quartiles, but are not longer than 1.5-times the IQR; data beyond these are shown as outliers (empty circles). Grassl.: grassland, Rob.: *Robinia* forest, Oak: oak forest, Pop.: poplar forest, Pine: pine forest.

3.4. Microclimate

Daily temperature cycles right above the soil surface followed very similar trends in silty-clayey and sandy sites (Fig. 6). At the end of winter, grasslands had a slightly more prominent peak during the day than forests, while the temperature difference was more apparent in spring and summer.

Table 2

Test results of the linear mixed-effects models of herb layer biomass and litter density in the two soil types. Significant results ($p < 0.05$) are marked with boldface. Pairwise comparisons were prepared only in models where habitat type had a significant effect. Grassl: grassland, Oak: oak forest, Rob: *Robinia* forest, Pine: pine forest, Pop: poplar forest.

Silty-clayey	Biomass		Litter	
	χ^2	p	χ^2	p
Model	79.04	<0.001	85.3	<0.001
	<i>t ratio</i>	p	<i>t ratio</i>	p
Grassl - Oak	-12.12	<0.001	12.84	<0.001
Grassl - Rob	-2.96	0.005	4.37	<0.001
Oak - Rob	9.03	<0.001	-8.48	<0.001
Sandy	Biomass		Litter	
	χ^2	p	χ^2	p
Model			73.58	
	<i>t ratio</i>	p	<i>t ratio</i>	p
Grassl - Pine	3.33	0.002	14.27	<0.001
Grassl - Pop	-2.55	0.017	9.24	<0.001
Grassl - Rob	3.35	0.017	5.18	<0.001
Pine - Pop	0.78	0.441	-4.80	<0.001
Pine - Rob	6.69	<0.001	-9.70	<0.001
Pop - Rob	5.89	<0.001	-4.12	<0.001

Autumn temperature values reflected the typical foggy or overcast weather of the season, having little diurnal fluctuation and negligible differences between habitat types. Night-time temperature patterns of grasslands and forests mostly overlapped among the studied sites. The only notable difference was that silty-clayey grasslands cooled down more in summer than the adjacent forests. Differences among forest types were apparently so subtle that they could hardly have any ecological effect.

4. Discussion

4.1. Soil moisture – tree cover relationship

In this study we aimed to reveal the importance of a so-far neglected environmental property, soil texture, in improving the spatial allocation of tree-based climate mitigation efforts. Our results indicate that the interplay among (i) the water consumption of the canopy and herb layers, (ii) precipitation interception by the canopy, herb and litter layers, and (iii) water conservation via canopy shading has different net effects on the vertical soil moisture dynamics, and thus on the groundwater recharge, on soils with contrasting texture. We found that topsoil and lower soil layer moisture regimes are mostly decoupled in both soil types, but processes in both layers can affect eventual deep percolation and thus groundwater recharge. Topsoil moisture content seems to be principally determined by its exposure to insolation (i.e. shaded by forest canopy or not), which has a drying effect via increasing temperature during the day (Zellweger et al., 2020). As a result, the topsoil of sandy poplar and *Robinia* and silty-clayey oak forests remained moist under the shade of the canopy during warmer months, while in grasslands the air got warmer, facilitating evaporative loss, and this was not counterbalanced by other effects of the vegetation, such as the lower transpiration and precipitation interception of grasslands compared to forests. Pine on sand as well as *Robinia* on silty-clayey soil, however, had as dry topsoil in summer as grasslands. In pine, we explain this with the interception of the thick litter layer and the permanently high canopy interception, while in *Robinia* forests the reason can be the very high herb layer evapotranspiration coupled with little canopy shading early in the season.

Moisture from precipitation that is supposed to supply deep soil moisture reserves needs to cross the topsoil, which has variable antecedent moisture content, as shown above. In sandy grasslands and pine forests, some moisture is thus lost to reach saturation to field capacity but in the other sandy forests less moisture is lost. On silty-clayey grasslands, however, the dry topsoil is easily bypassed as the topsoil cracks when it gets dry, opening channels for percolation to deeper layers (Smith et al., 2018; Qi et al., 2020). This excess water is clearly indicated by our data, as in summer the topsoil of grasslands was dry but the deeper soil turned wetter than it had been at the end of spring. Conversely, no excess water appeared in the deeper layers of silty-clayey *Robinia* forests in summer, as the dry topsoil of *Robinia* does not crack, probably due to its lower compaction compared to our grazed and thus trampled grasslands (cf. Valente et al., 2021), and thus summer rains are retained, and eventually lost to evapotranspiration.

The overall effect of vegetation cover on the moisture content in the deeper layers was mostly the opposite to those in the topsoil. Grasslands had little or no effect on the moisture level below 40 cm, that is, evapotranspiration by herbaceous plants and the increased evaporation due to the warmer microclimate did not affect moisture content below the very top of the soil. Although the maximum rooting depth of many temperate grassland species can be rather high (Canadell et al., 1996), the overwhelming majority of their roots are in the topsoil (Schulze et al., 1996; Poelplau et al., 2019). In contrast, forests on sand had clear negative effects on deep soil moisture, and the strength of the effect depended on the annual canopy lifetime. Pine stripped deeper soil layers from their moisture content for most of the year, while poplar and *Robinia* did so for shorter periods. Thus, the moisture that could seep through the topsoil can hardly make it to the groundwater, lying several meters below the soil surface. In contrast, oak forests on silty-clayey soil had stable deep soil moisture

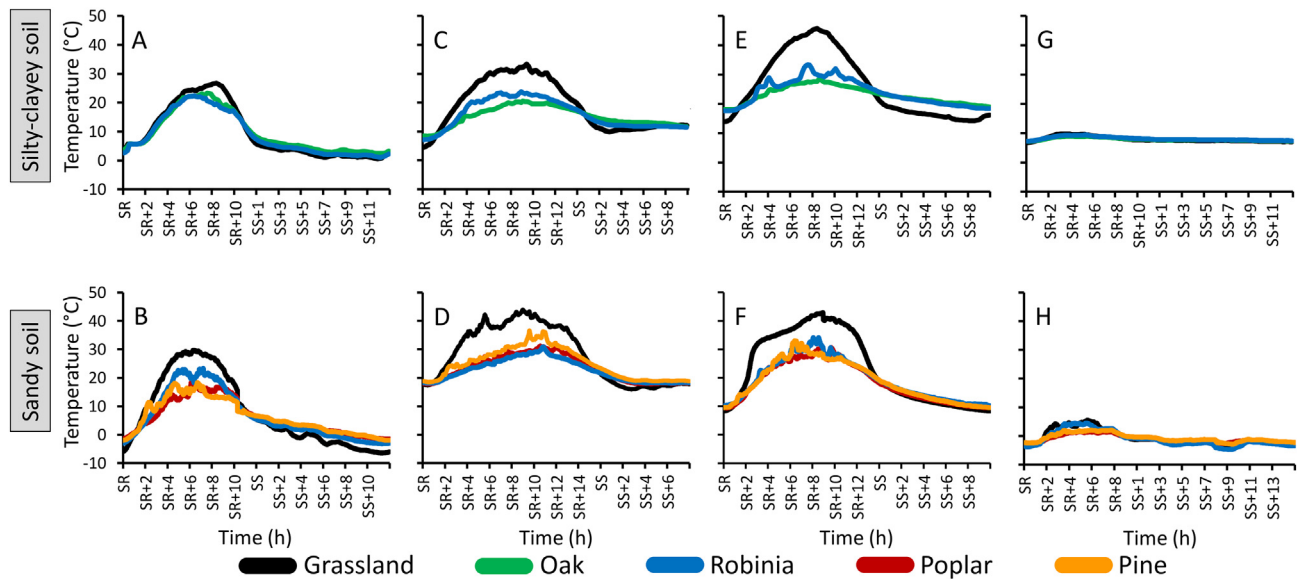


Fig. 6. Air temperature right above the soil surface of the studied habitats sites at the end of winter (A, B), spring (C, D), summer (E, F) and autumn (G, H). SR: sunrise, SS: sunset. Each curve shows the average of records from four sensors.

conditions throughout the year. Their moisture content did not rise notably in winter months either, although they received the same amount of precipitation as grasslands and *Robinia*, and had no higher rate of interception, as oak is deciduous. The stability of water content may be explained by a better deep infiltration, which this native species can potentially support due to an improved soil life compared to the non-native *Robinia* (Lazzaro et al., 2018).

4.2. Restoration dilemma: target forest or grassland?

Considering the combined effects of vegetation cover on top and deeper soil layers, we conclude that afforestation in the coarse-textured sandy soils heavily limits the rate of groundwater recharge (see also Huang et al., 2020 for similar findings in a sandy region of China). In line with this, afforestation is now a more and more widely accepted contributor of recent groundwater decline in the Kiskunság Sand Ridge (Tölgyesi et al., 2021; Szabó et al., 2022). In addition, groundwater decline in afforested sandy landscapes can cause water shortage in lower lying, groundwater-fed wetlands and negatively affect agricultural production that relies on irrigation from water reserves of the soil. Afforestation may yield higher carbon gain than grassland restoration due to the low soil organic carbon content of dry sandy soils (Plante et al., 2006), but farther away, the water deficit may offset the carbon gain by lowering plant biomass production, and desiccating wetlands and turning their carbon-rich soil from sink into source, as seen globally during wetland desiccation (Armentano and Menges, 1986; Hooijer et al., 2010; Zhang et al., 2020). Afforestation above the historical proportion of forest cover in sandy dryland is thus clearly advised against, and grassland restoration is preferred. However, if forests are still to be planted, they should be selected from the native species pool and/or species with short canopy lifetime should be preferred. The typical amount of herbaceous biomass or litter amount of the forest type is not necessarily an important factor to consider as their effect seems to be low compared to the tree canopy traits.

On fine textured soil, like in our silty-clayey sites, the situation is not so black-and-white. Deeper soil moisture is less compromised by trees and thus groundwater recharge seems less hindered in silty-clayey forested sites than in sandy ones. In soil types where crack-formation is not typical in dry periods, or if the precipitation peak does not coincide with the cracked period, the advantage of grasslands in recharging groundwater resources further decreases. Restoration practitioners may thus put somewhat more emphasis on forests or other wooded rangelands (such as wood-

pastures; Bergmeier et al., 2010) on fine-grained soils. Trees with long annual canopy life-time are not advised against, since the shading effect to retain moisture is more important on silty-clayey soil than on sand because the moisture needs more time to seep into layers unexposed to the drying effect of insolation.

The slow rate of infiltration on fine textured soil can result in infiltration excess water after significant precipitation events, supporting overland flow (Wilcox et al., 2017). In flat landscapes, like the studied ones, water movements via overland flow are negligible, but in regions with more structured topography changes in overland flow may significantly affect downslope water supply (Jin et al., 2020), potentially requiring adjustments on forest vs. grassland restoration. There is copious literature that overland flow and hence stream flow at the bottom of catchments decreases with increasing forest cover in topographically structured temperate environments (Farley et al., 2005; Khorchani et al., 2022). So, the decision on whether or not to plant trees in such areas should be based on watershed-scale needs to avoid flash floods vs. the need for downstream water supply: The former supports forests (e.g. Jin et al., 2020), while the latter grasslands (e.g. Buytaert et al., 2007). Sandy areas in water-limited environments have little infiltration excess water to feed overland flow (Knighton et al., 2020), and our studied sandy region in Hungary also completely lacks natural streaming water courses.

4.3. Adjusting forest vs. grassland restoration needs to present conditions

Interestingly, when comparing soil and forest cover maps of Hungary, the opposite pattern can be seen compared to our recommendation, as water-limited sandy regions are more forested than the ones on fine-textured soils (Fig. 7). In many regions within the temperate forest-steppe belt, high forest cover actually delineates sandy deposits, such as the Deliblato Sands in Serbia (Milenković et al., 2017), or the dry alluvial sand terraces of Ukraine (e.g. Bujanov, 2014), whereas the surrounding non-sandy areas almost completely lack woody vegetation (Fig. 8). What is the reason for this pattern?

Since moisture from precipitation remains within reach for herbaceous species (crop and native grassland species alike) for a longer period in fine textured soil than in sand (Acharya et al., 2018), crop production, as well as grassland use (grazing or mowing) can be the preferred land use types, leaving little room for forestry. In contrast, moisture can seep down to lower layers on sandy soil faster, where the deeper roots of woody species can take advantage of it, and, as our findings suggest, can very effectively use

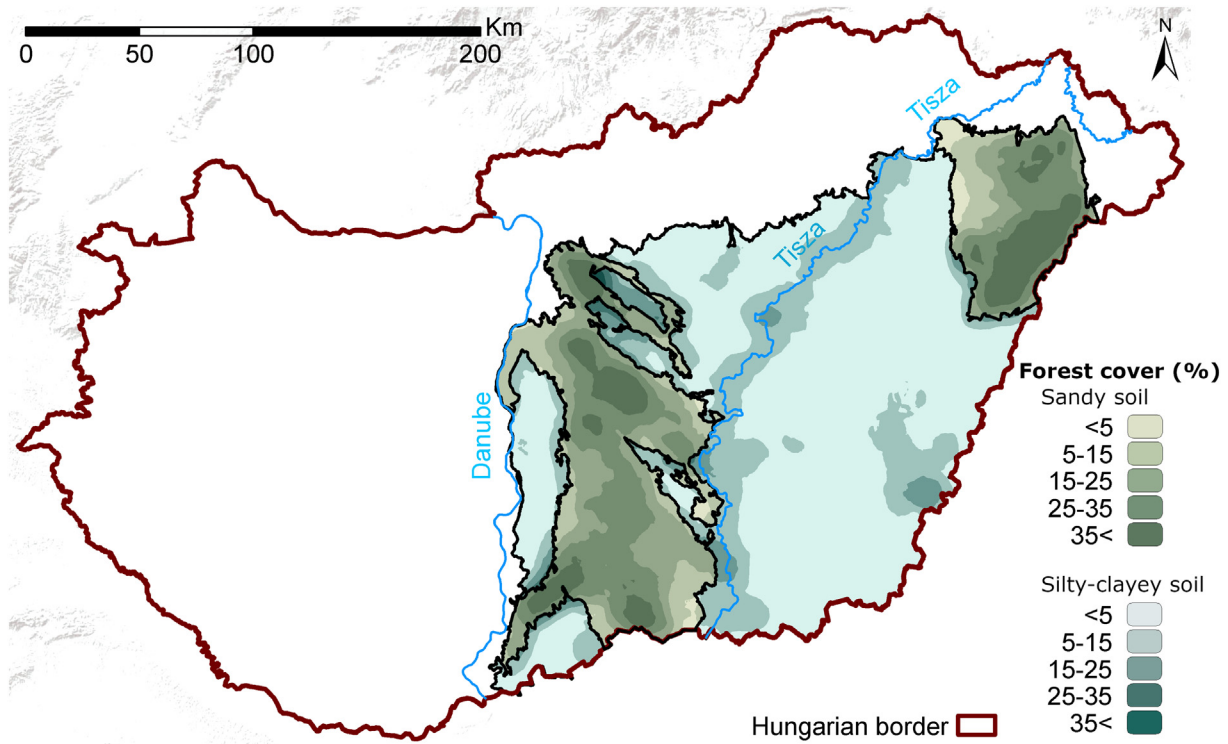


Fig. 7. Forest cover on sandy and silty-clayey regions of the main forest-steppe zone of the middle and eastern parts of the Great Hungarian Plain. Forested vegetation is concentrated on sandy soil, while it is much scarcer on finer textured soil and occurs mostly along a major river. Soil type contours are based on the soil map of Hungary by Várallyay (1985), and tree density is derived from the Ecosystem Base Map of Hungary (Hungarian Ministry of Agriculture, 2019).

it up at the peak of the vegetation period. Furthermore, water-limited sandy areas are more erosion prone to grassland overuse, while forest cover can inhibit the erosion of sand soil (e.g. Li and Xu, 2019). Due to these reasons, it is no surprise that forest is a more preferred land use type on sandy than on fine textured soil. These potential land user preferences are also in good agreement with classical concepts of tree-grass interaction, such as the ‘niche separation by depth’ model (Scholes and Archer, 1997), which predicts competitive advantage for trees on sandy and disadvantage on fine textured soils in water-limited ecosystems because trees generally use

water from lower soil layers than herbaceous species (Rossatto et al., 2014). However, land users can often deviate from the natural proportions by exaggerating the preferences, and thus causing e.g., severe groundwater decline in sandy areas (Zeng et al., 2009; Tölgyesi et al., 2021), or the complete loss of forest habitats with all their ecosystem services and biodiversity on fine textured soils, as shown on Fig. 7.

The contrasting biases of land use on fine and coarse textured soils require opposite restoration activities, and these are now fully in line with our recommendations. On fine textured soil, there may be a large potential

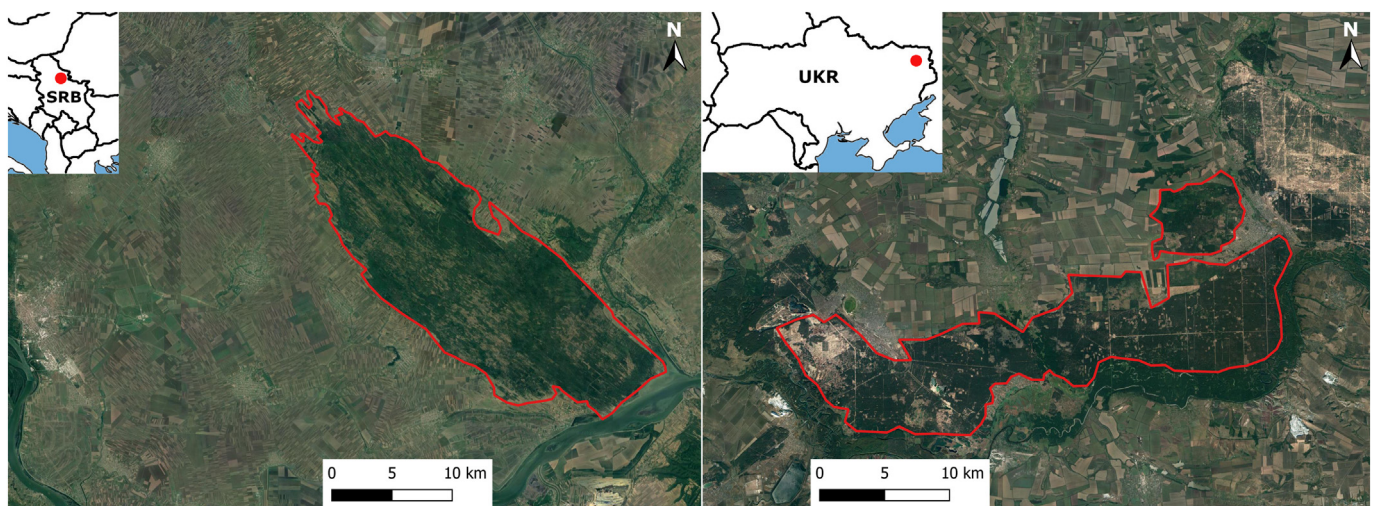


Fig. 8. Two over-forested dry sand regions (polygons with red margin) in the Eurasian forest-steppe belt. Both of them are surrounded by treeless, cultivated areas with finer textured soils, mostly loess. A: Deliblato Sands in Serbia, B: alluvial sand terraces near Severodoneck, Ukraine. The forested belt south of the dry sand area of the Ukrainian site is moist riverine forest; therefore, it is excluded from the polygon.

of forest restoration beside grassland restoration, while on sandy soils, the focus should be on grassland restoration. Many dry sand areas have been planted with exotic trees worldwide, e.g. around the North American Great Lakes (Leege and Murphy, 2001), in Patagonia (Hess and Austin, 2017) as well as in the presently studied Hungarian sand ranges. In these regions, forest restoration (functionally afforestation) should avoid presently treeless sites, but focus on increasing the naturalness of forested sites, because plantation forests host little biodiversity (Rédei et al., 2020) and their carbon sequestration capacity is also a fraction of natural forests (Lewis et al., 2019).

5. Conclusions

We showed that soil texture strongly interacts with the effect of canopy, herb and litter layer interception, canopy and herb layer transpiration, and microclimate on the water budget of water-limited temperate regions, and this has fundamental consequences for the utility of large-scale tree-based climate mitigation strategies. We emphasize that increasing forest cover in dry sandy regions is advised against because it can result in a severe negative water balance and a reduced net carbon gain at the watershed-scale. However, on fine textured soil, the desiccating effect is less pronounced; therefore, forest restoration has better prospects from the hydrological viewpoint. Opportunities for the implementation of our recommendations are ensured by current and historical land use preferences, which resulted in a vast quantity of marginal and abandoned croplands on fine-textured soil, which are suitable for the restoration of forests and wooded rangelands. At the same time many sandy areas have already undergone excessive afforestation, potentially making it easier to avoid further afforestation and to focus on grassland restoration as well as the improvement of the naturalness of plantation forests.

CRedit authorship contribution statement

Csaba Tölgyesi: Conceptualization, Data collection, Data analysis, Writing original draft.

Alida Anna Hábcenyus: Data collection, Data analysis, Writing-Reviewing and Editing.

András Kelemen: Data collection, Writing- Reviewing and Editing.

Péter Török: Writing. Reviewing and Editing.

Orsolya Valkó: Data collection, Writing- Reviewing and Editing.

Balázs Deák: Data collection, Writing- Reviewing and Editing.

László Erdős: Data collection, Writing- Reviewing and Editing.

Benedek Tóth: Data collection, Data analysis, Writing- Reviewing and Editing.

Nándor Csikós: Graphical illustration, Writing- Reviewing and Editing.

Zoltán Bátorja: Data collection, Writing- Reviewing and Editing.

Data availability

Data will be archived in the Dryad Data Repository upon the acceptance of the paper.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Csaba Tölgyesi reports financial support was provided by National Research Development and Innovation Office. Laszlo Erdos reports financial support was provided by National Research Development and Innovation Office. Zoltan Batorja reports financial support was provided by National Research Development and Innovation Office. Balazs Deak reports financial support was provided by National Research Development and Innovation Office. Orsolya Valko reports financial support was provided by National Research Development and Innovation Office. Peter Torok reports financial support was provided by National Research Development and Innovation Office.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2022.158960>.

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