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The effects of tree establishment on water and salt dynamics in naturally salt-affected grasslands

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Abstract Plants, by influencing water fluxes across the ecosystem–vadose zone–aquifer continuum, can leave an imprint on salt accumulation and distribution patterns. We explored how the conversion of native grasslands to oak plantations affected the abundance and distribution of salts on soils and groundwater through changes in the water balance in naturally salt-affected landscapes of Hortobagy (Hungary), a region where artificial drainage performed ~150 years ago lowered the water table (from –2 to –5 m) decoupling it from the surface ecosystem. Paired soil sampling and detailed soil conductivity transects revealed consistently different salt distribution patterns between grasslands and plantations, with shallow salinity losses and deep salinity gains accompanying tree establishment. Salts accumulated in the upper soil layers during pre-drainage times have remained in drained grasslands but have been flushed away under tree plantations (65 and 83% loss of chloride and sodium, respectively, in the 0 to –0.5 m depth range) as a result of a five- to 25-fold increase in infiltration rates detected under plantations. At greater depth, closer to the current water table level, the salt balance was reversed, with tree plantations gaining 2.5 kg sodium chloride m^{-2} down to 6 m depth, resulting from groundwater uptake and salt exclusion by tree roots in the capillary fringe. Diurnal water table fluctuations, detected in a plantation stand but not in the neighbouring grasslands, together with salt mass balances suggest that trees consumed ~380 mm groundwater per year, re-establishing the discharge regime and leading to higher salt accumulation rates than those interrupted by regional drainage practices more than a century ago. The strong influences of vegetation changes on water dynamics can have cascading consequences on salt accumulation and distribution, and a broad ecohydrological perspective that explicitly considers vegetation–groundwater links is needed to anticipate and manage them.

Keywords Ecohydrology · Salinization · Afforestation · Groundwater use · Drainage

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Introduction

Plants, through their contrasting capacity to access, transport, and evaporate soil moisture, exert a strong influence on water dynamics, with vegetation changes translating into hydrological shifts that manifest at stand, landscape, or regional scales. Because the dynamics of salts are intimately linked to water transport across the ecosystem–vadose zone–aquifer continuum, vegetation changes have also the potential to alter salt accumulation and distribution patterns at multiple scales. Land use changes involving transitions between grass- and tree-dominated covers have some of the strongest effects on evapotranspiration, runoff and deep drainage patterns, having the potential to redistribute salts. In this paper, we explore how the conversion of native grasslands to tree plantations affects the abundance and distribution of salts in the ecosystem–vadose zone–aquifer continuum of naturally salt-affected landscapes in the sedimentary plains of Hortobagy (Hungary), focusing on the patterns and mechanisms of water and salt transport shifts.

The effects of vegetation changes on salt accumulation often vary according to the spatial and temporal scale considered. For example, land use changes causing increased deep drainage, such as the replacement of dry forest with crops or pastures, can remove local soil salts through flushing in a few years (Scanlon et al. 2005). In the longer term, however, the same land use change can facilitate salt transport to aquifers that, due to the increased recharge and rising water levels, may transport and release salts back to the surface in a low topographic position (George et al. 1997). This situation has been observed in Western and Southern Australia and the Sahel (Culf et al. 1993; Wood 1924) where the scale of vegetation changes has been regional.

While most of the knowledge about the effects of vegetation change on salt dynamics derives from deforestation situations, the reverse change, namely grassland afforestation, has also the potential to modify salt dynamics through at least two mechanisms: higher evapotranspiration rates driving the onset of groundwater discharge and salt build up and increased soil macroporosity enhancing water infiltration and salt flushing. Compared with grasses, trees often present higher evaporative capacity, dictated by greater aerodynamic conductance and deeper root systems (Canadell et al. 1996; Kelliher et al. 1993). When this greater evapotranspiration capacity occurs through groundwater use, tree establishment on grasslands can trigger groundwater and soils' salinization (Heuperman 1999; Jobbágy and Jackson 2004). By means of higher root density, litter amount and mesofaunal activity, tree sites often exhibit better soil physical properties for water movement in comparison with grasslands (Devitt and Smith 2002; Dunkerly 2000). When this soil improvement translates to an enhanced downward movement of water, decreased runoff, increased infiltration and salt flushing are expected results of tree establishment (Eldridge and Freudenberger 2005; Mishra et al. 2004).

Naturally salt-affected grasslands occupy sedimentary plains worldwide. These ecosystems are located in humid and sub-humid climates and include regions such as the Great Plains of western Canada (FAO 1991), Pampa Deprimida and Bajos Submeridionales in central Argentina (Lavado and Taboada 1988; Morras and Candioti 1982), the Carpathian basin (Hungary, Romania, Serbia and Slovakia) (Toth and Rajkai 1994) and the lowlands of western Siberia (Bazilevich 1965). Although rainfall, often exceeding the evapotranspiration outputs of grassland communities, could be enough to flush accumulated salts in these regions, low slope gradients leave water excesses within the region. This surplus water often causes shallow groundwater levels, favouring local discharge through plant transpiration, soil evaporation, or pond evaporation in the lowest landscape positions, with all these water evacuation pathways leading to salt accumulation. Reclamation measures aimed at interrupting the discharge regime by lowering the water table and channelling excess water to large rivers, as happened in Hungary ~150 years ago, have proved to be insufficient to solve the problem, because the poor soil physical properties hinder the salt leaching process (Schofield et al. 2001; Szabolcs 1989). Tree establishment in these situations could lead to contrasting outcomes, modifying water flows and processes operating in opposite ways. On the one hand, trees, through their greater rooting depth compared to grasses (Canadell et al. 1996), could re-establish the discharge regime interrupted by past reclamation leading to deeper salt accumulation (see Heuperman 1999; Jobbágy and Jackson 2004) but, on the other hand, the improved soil physical conditions generated by trees (Devitt and Smith 2002) could promote salt leaching from the upper soil layers (see Mishra et al. 2004).

In this study we explored how the establishment of oak plantations (*Quercus robur* L.) on naturally salt-affected grasslands of Hortobagy (Hungary) affected the distribution of salts in soils and groundwater. We linked shifts in salt and water dynamics based on the following hypotheses: (1) tree plantations, through the improvement of soil physical conditions, increase infiltration rate and promote leaching of salts from the upper soil layers; (2) the deeper root systems that tree plantations have in comparison with grasslands, result in groundwater use and salt accumulation in deep soil layers and aquifers. We explored these hypotheses across seven grassland–tree plantation pairs, combining salinity measurements based on electromagnetic induction techniques and soil coring with measurements of infiltration and pan evaporation rates as well as water table depth and soil water content records.

Materials and methods

The region

The Great Hungarian Plain (Fig. 1), covering 47,000 km^2 of eastern Hungary, is a Quaternary sedimentary basin containing eolian and fluvial sediments where the extremely flat topography and the poor drainage network determine the presence of a shallow groundwater across most of the area (Szabolcs 1989). Before the hydrological

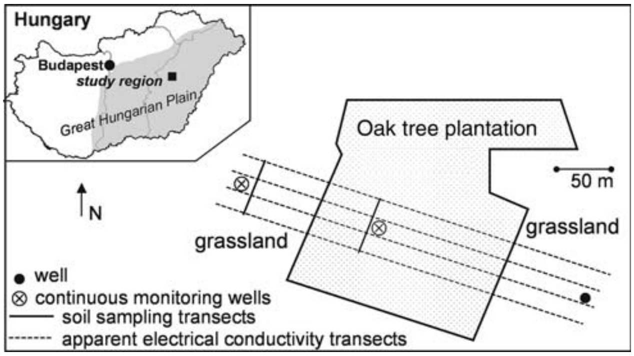

Fig. 1 Study region showing the Great Hungarian Plain and the oak plantation and grassland pair studied in greater detail (*Site E*). Soil sampling was performed in transects parallel to the plantation edge. Apparent electrical conductivity was assessed in four transects crossing the plantation. Water table levels were monitored continuously during a 2.5-month period at two wells

regulation of the region, considerable parts of the Plain were permanently or temporarily inundated (Ábrahám and Bocskai 1971). These conditions, together with the poor soil hydrophysical properties, favoured the development of salt-affected soils, mainly solonetz and solonchak types (Schofield et al. 2001). The native vegetation of the area is dominated by grasses and the mean annual temperature and precipitation approach 10.2°C and 540 mm, respectively (Béla 1972; Toth et al. 1991).

In the nineteenth century, artificial drainage operations were initiated across the Great Hungarian Plain and involved the regulation of the Danube and Tisza rivers and the construction of drainage channels towards them. Although this large operation led to a significant decline of groundwater levels, the results were not completely satisfactory because, in most places, salts still remain in the soil profile as a result of poor soil physical conditions preventing their leaching (Schofield et al. 2001; Szabolcs 1989). Soil properties here constrain land use options to the raising of livestock on native pastures (Ábrahám and Bocskai 1971). However, several tree plantations have been established during the last century as an alternative use of these soils, providing a useful setting to explore the effects of grassland afforestation on water and salt dynamics.

Study sites

The study sites were located close to the city of Püspökladany (47°20′N, 21°06′E) in the Hortobágy National Park (Hungary). To explore the effects of tree establishment on salt and water dynamics of naturally salt-affected grassland, we carried out a space-for-time substitution approach (Pickett 1989) selecting seven oak plantation—grassland pairs (sites A–G). The implicit assumption in this approach is that the initial edaphic variations between paired stands before tree establishment were negligible. Several factors provide support to this assumption warranting careful comparisons. Afforested stands were planted following fences and property limits (I. Csiha, personal communication) and their establishment was not biased towards a particular landscape unit or topographic position. A detailed topographic map of the area and direct observations in the field confirmed that both stands in each site were located at the same topographic situation. The differences in topographic levels between sampling sites were very small (<0.25 m), which is of particular importance for this region considering that groundwater depth played a dominant role in the development of these soils (Szabolcs 1989; Toth et al. 1991). In addition, soil sampling transects in paired stands were separated by less than 80 m, which, together with the homogeneity of the loessic sediments of the Hortobagy region (Ábrahám and Bocskai 1971), hinders strong initial pedogenic variation across stands.

Selected plantations were 2–5 ha in size and were planted between 1930 and 1965. No fertilization or irrigation were ever applied and plantations were never thinned or pruned. Current oak density ranged from ~450 to ~800 trees ha^{-1} and basal area from 25 to 40 m^2 ha^{-1}. The understory was sparse and dominated by *Fraxinus* sp. Grassland stands were dominated by *Alopecurus pratensis*, *Agropyron repens*, *Limonium gmelini* and *Festuca pseudovina* and were neither fertilized nor irrigated. At three sites (E, F and G), soil texture contributed to an evaluation of the initial soil homogeneity between paired stands since it was unlikely to change in the time span of this study (Jenny 1941). Sand, silt and clay contents (hydrometer method; Bouyoucos 1962) did not differ significantly between tree and grassland stands at any given site ($P > 0.10$ in most cases, $n = 3$; Table 1). However, some significant differences were observed across sites ($P < 0.10$, $n = 2$; Table 1). The pattern of soil texture was characterized by a silty loam texture in shallow soil layers (0–10 cm depth) and a silty clay one in deeper horizons with the clay content reaching ~50% (Fig. 2).

Salt dynamics

We characterized salt accumulation patterns in two complementary ways. Surface salinity (<1.5 m) was assessed using electromagnetic induction techniques in the seven oak plantation–grassland pairs and in a subset of three of these sites a detailed soil characterization was performed up to a depth of 3 m based on soil sampling. In each stand >15 electromagnetic measurements were taken. In site E, four parallel transects crossing the plantation were established (Fig. 1). Measurements were taken every 5 m close to the plantation–grassland edges and every 10 m in the centre of the plantation. A satisfactory correlation was

Table 1 Evaluation of soil texture[a] differences across vegetation types and sites based on clay, silt and sand contents of eight layers. The probability values of the *t*-test (vegetation effect within site, $n = 3$) and ANOVA (site effect, $n = 2$) are indicated

Depth (cm)	*P*-value of vegetation effect									*P*-value of site effect		
	Clay			Silt			Sand			Clay	Silt	Sand
	Site E	Site F	Site G	Site E	Site F	Site G	Site E	Site F	Site G			
0–10	0.86	0.78	0.70	0.61	0.11	0.44	0.67	0.13	0.32	0.001	0.052	0.17
10–30	0.69	0.30	0.36	0.56	0.50	0.12	0.09	0.64	0.59	0.14	0.44	0.65
30–50	0.42	0.89	0.97	0.45	0.65	0.43	0.60	0.20	0.16	0.006	0.17	0.10
50–100	0.60	0.67	0.57	0.73	0.49	0.17	0.35	0.94	0.08	0.02	0.04	0.29
100–150	0.57	0.51	0.12	0.51	0.98	0.18	0.35	0.16	0.23	0.04	0.03	0.14
150–200	0.71	0.48	0.49	0.74	0.66	0.18	0.07	0.09	0.52	0.008	0.001	0.83
200–250	0.42	0.42	0.24	0.14	0.53	0.10	0.04	0.63	0.43	0.015	0.15	0.38
250–300	0.71	0.18	0.16	0.34	0.19	0.14	0.11	0.64	0.95	0.04	0.10	0.04

[a] Soil particle size distribution was determined by the hydrometer method (Bouyoucos 1962)

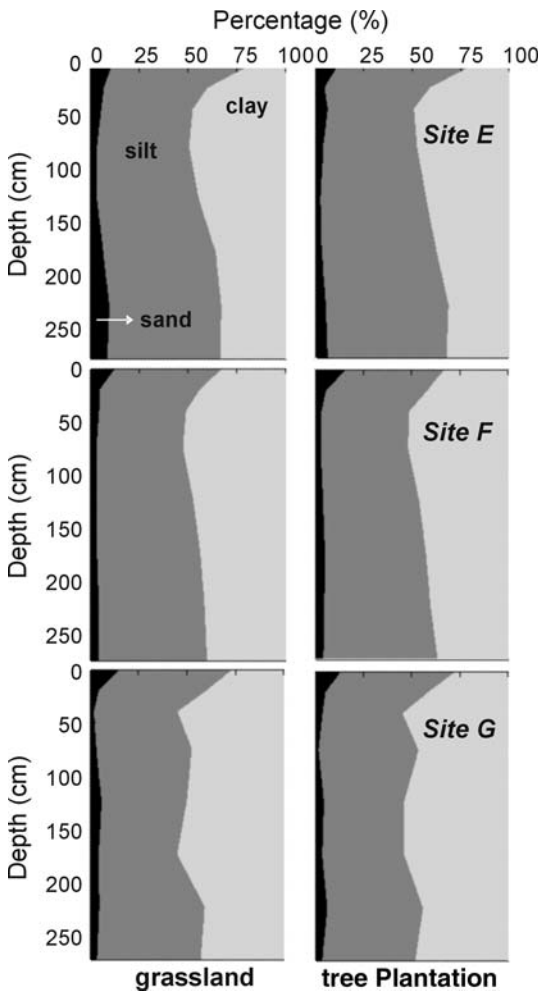

Fig. 2 Soil particle size distribution through the soil profile for three sites (*Sites E*, *F* and *G*). Soil texture (sand, silt and clay %) was determined by the hydrometer method (Bouyoucos 1962) ($n = 3$ per stand)

obtained between electromagnetic measurements and soil conductivity measured in a 1:2.5 soil–water extract ($r^2 = 0.91$, $P<0.0001$, $n = 24$).

At sites E, F and G, mineral soil was sampled at depth intervals of 0–10, 10–30, 30–50, 50–100, 100–150, 150–200, 200–250, 250–300 cm. At site E, two soil cores (one in the grassland and one in the plantation) were taken down to a depth of 6 m and soil sampled at 50-cm intervals. At each stand we randomly located four soil pits along a transect parallel to the grassland–plantation edge. Electrical conductivity, pH, chloride, and sodium content were determined in a 1:2.5 soil–water extract. Soil texture, determined by the hydrometer method (Bouyoucos 1962), was used to give a rough estimate of bulk density (Stefanovits 1981). Groundwater was sampled for electrical conductivity, pH, chloride and sodium analysis in a three-point transect crossing the plantation at site E (Fig. 1). Electrical conductivity was measured with a conductivity meter (Orion model 115; Orion Research, Mass.). Sodium and chloride concentrations were determined with ion-selective electrodes (Frankenberger et al. 1996; Talibudeen 1991). The content of carbonates was assessed using Scheibler's method (Williams 1948) and the pH with an electrode (Thomas 1996).

Water dynamics

We characterized soil moisture, soil water infiltration, pan evaporation and water table levels. Gravimetric water contents and ponded infiltration rates were determined at the same sites where soil sampling was done (sites E–G), while evaporation rates and water table dynamics were assessed only at site E. Infiltration measurements were performed using double-ring infiltrometer (615–805 cm^2

inner area of rings, $n = 3$) (Ahuja et al. 1976). We also performed pan evaporation measurements for 10 days (29 August 2005–6 September 2005) using micro-evaporimeters (10 cm diameter, $n = 3$) placed on the ground (Feng et al. 2004).

We examined diurnal fluctuations in groundwater levels as an indicator of groundwater use by the vegetation (Engel et al. 2005; Loheide et al. 2005; White 1932). For this purpose, we measured water table levels during a 72-day period covering the leaf fall stage (30 August 2005–9 November 2005). Two permanent monitoring wells were located 40 m from the plantation–grassland edge (see Fig. 1). We registered groundwater levels at high vertical (<3 mm) and temporal (30 min) resolutions using pressure transducers (model PDCR 830; Druck, Leicester, UK) connected to dataloggers (Level Tracker WLS-2109e; Telog Instruments, Rochester, N.Y.).

In order to estimate groundwater use by the vegetation we used two independent approaches: diurnal watertable fluctuations and chloride mass balance. We used the method proposed by Engel et al. (2005) where direct withdrawals from groundwater through transpiration (E_{gw}, mm day^{-1}) are calculated as:

$$E_{\mathrm{gw}} = \left(24\,\delta z/\delta t - \delta z_{\mathrm{ref}} + \delta z_{\mathrm{pl}}\right) \times \mathrm{SY}$$

where $\delta z/\delta t$ is the rate of increase in groundwater levels (mm h^{-1}) from midnight to 4 a.m. multiplied by 24 to estimate inputs from the shallow aquifer in the surrounding grassland (Darcian flow), δz_{ref} represents regional groundwater fluctuations not associated with local influences and is estimated from net daily changes (m day^{-1}) in the reference monitoring well in the grassland, δz_{pl} is the net change in plantation groundwater level over a 24-h period, and SY is the aquifer specific yield (m mm^{-1}), estimated as 0.026 according to Loheide et al. (2005) for silty sediments. The chloride balance approach is based on the conservative behaviour of this element (Walker 1998) and the annual average groundwater uptake (aE_{gw}) by the vegetation is calculated as:

$$aE_{\mathrm{gw}} = \frac{\Delta \mathrm{Cl}_{\mathrm{soil}}}{[\mathrm{Cl}]_{\mathrm{gw}} \times n}$$

where, aE_{gw} (mm year^{-1}) is the annual average net groundwater uptake rate (E_{gw}–deep drainage) over the plantation lifespan ($n = 43$ year), Δ $\mathrm{Cl}_{\mathrm{soil}}$ (g m^{-2}) is the plantation–grassland difference in total soil chloride down to the bottom of the rooting zone (6 m in this study) and $[\mathrm{Cl}]_{\mathrm{gw}}$ (g l^{-1}) is the groundwater chloride concentration below the rooting zone.

Results

Salt dynamics

Soils under grasslands and tree plantations displayed consistently different salt distribution patterns, with shallow salinity losses and deep salinity gains accompanying tree establishment. Salinity levels in the upper ~1.5 m of the soil, assessed by electromagnetic induction, were significantly higher under grassland stands throughout the seven grassland–oak plantation pairs considered (2.2-fold on average, ranging from 1.3- to threefold) (Fig. 3a). Conductivity transects performed at site E, showed a steep salinity change near the plantation edge and as close as 10 m from the plantation differences already became significant (1.36 and 0.47 dS m^{-1} for grassland and plantation, respectively; $P < 0.01$, paired *t*-test) (Fig. 3b).

In agreement with electromagnetic induction measurements, soil profiles showed higher electrical conductivity values in the upper soil layers of grassland stands, with differences being more evident at sites E and F (Fig. 4). Integrating the upper 2 m of the profile, the electrical

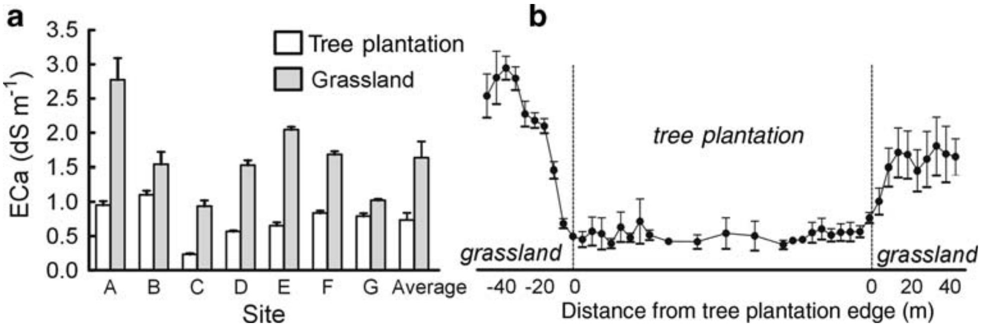

Fig. 3a, b Apparent electrical conductivity (*ECa*) measurements performed in seven plantation–grassland pairs. Mean values and SD of each site, as well as the mean value of the seven sites, are indicated (**a**). Differences between and within sites were significant in all cases ($P < 0.05$). Mean value and SD of four transects crossing site E are shown (**b**). Measurements were taken every 5–10 m

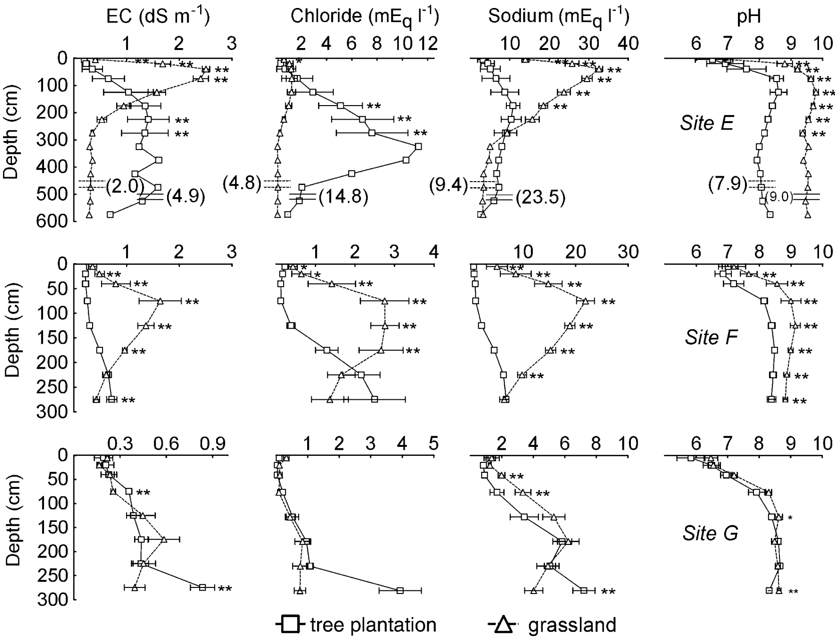

Fig. 4 Soil properties under native grassland and oak plantations pairs at three sites. Electrical conductivity (*EC*) (dS m^{-1}), chloride (mEq l^{-1}), sodium (mEq l^{-1}) and pH were measured in a 1:2.5 soil–water extract (mean ± SE, $n = 4$ for 0–3 m). At site E, two soil cores were taken down to a depth of 6 m ($n = 1$ for 3–6 m). The levels of the watertable during the measurement period are indicated by *horizontal lines* and their corresponding measured values are *in parentheses*. *$P < 0.10$, **$P < 0.05$, between stands

conductivity of grassland soils doubled, on average, the conductivity of plantation soils (1.06 and 0.50 dS m^{-1} for grasslands and plantations, respectively; $P < 0.10$, paired *t*-test). However, the pattern of soil conductivity reversed at deeper soil layers, with plantations showing the highest values (Fig. 4). Plantation soils had 2.5 times higher conductivity than grassland soils in the 2.5- to 3-m layer ($P < 0.05$, paired *t*-test, Fig. 4).

Chloride and sodium profiles followed electrical conductivity patterns with grassland stands showing higher concentrations in shallower soil layers and plantations in deeper soil layers (Fig. 4). Integrating the upper 50 cm of soil profiles, grassland soils stored 2.8 times more chloride (40.7 and 14.3 g m^{-2} for grasslands and plantations, respectively; $P < 0.05$, paired *t*-test) and 6.1 times more sodium (478.1 and 77.9 g m^{-2} for grasslands and plantations, respectively; $P < 0.10$, paired *t*-test) than plantation ones. In the 2.5- to 3-m soil interval, plantations had 5.6 times more chloride than grasslands (48.8 and 275.6 g m^{-2} for grasslands and plantations, respectively; $P < 0.05$, paired *t*-test) whereas differences in sodium storage between both types of cover disappeared at this depth (256 and 291.3 g m^{-2} for grasslands and plantations, respectively; $P = 0.23$, paired *t*-test). Although the pattern at site G was not as clear, it also followed the general trend (Fig. 4) with shallow salinity loss (60 and 40% loss of chloride and sodium, respectively, in the 0- to 0.5-m-depth range) and deep salinity gain (67 and 27% gain of chloride and sodium, respectively, in the 2.5- to 3-m-depth range). Soil profiles of calcium carbonate showed larger variability and the integration of the complete soil profile (0–3 m) resulted in higher accumulation at grassland stands (245.2 and 158.9 kg m^{-2} for grasslands and plantations, respectively; $P < 0.10$, paired *t*-test).

Soil pH under grassland and tree plantations showed significant contrasts throughout the profile, with the maximum difference reaching ~1.1 pH units at 30–50 cm depth (Fig. 4) (8.3 and 7.2 for grasslands and plantations, respectively; $P < 0.10$, paired *t*-test). Averaged over the complete soil profile (0–3 m), soil pH was ~0.7 unit lower under plantations than grasslands (8.82 and 8.13 for grasslands and plantations, respectively; $P < 0.10$, paired *t*-test). The groundwater sampling transect at site E (see Fig. 1) showed saltier groundwater at the tree plantation in comparison with the two grassland wells. In the plantation, electrical conductivity approached 4.9 dS m^{-1} at the water table and chloride and sodium concentrations 14.8 and 23.5 mEq l^{-1}, respectively. In the surrounding grasslands the salt content of the groundwater decreased substantially, as evidenced by electrical conductivities averaging 2 dS m^{-1} and chloride and sodium content approaching 4.8 and 9.4 mEq l^{-1}, respectively.

Water dynamics

We identified contrasting patterns of soil moisture storage, infiltration, pan evaporation, and water table-level dynamics between grasslands and oak plantations. Moisture profiles showed significantly drier soils in plantations compared to grassland (Fig. 5), suggesting higher water use in the former. On average, soil profiles had ~3% higher gravimetric water content under grasslands than tree plantations (18.5 vs. 15.7%; $P < 0.05$, paired *t*-test). Differences in the total amount of water stored down to 3 m

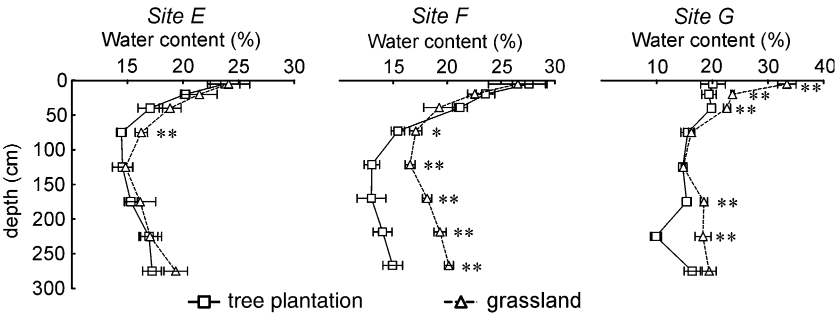

Fig. 5 Water content profiles under native grassland and oak plantations pairs at three sites (mean ± SE; $n = 4$ per stand). $^{*}P < 0.10$, $^{**}P < 0.05$ between stands

depth ranged from 44 mm (site E; 662 vs. 706 mm) to 139 mm (site F, 626 vs. 765 mm), averaging ~100 mm. Plant-available water (difference between observed water storage and that expected from a –1.5 MPa soil water potential contents; Campbell 2003) decreased ~74 mm (141 vs. 215 mm) after afforestation, with site F showing the greatest difference (139 mm).

Tree plantations had higher infiltration rates and lower evaporative demand at the ground compared with grasslands. Averaging the three sites, the infiltration rates obtained from the double-ring infiltrometer were 15 times greater under plantations ($P < 0.05$, paired t-test), with absolute differences ranging between 12 (site E) and 70 mm h^{-1} (Site F) (Fig. 6). Evaporative demand at the ground was 5 times higher in the grassland stand (site E, Fig. 6, $P < 0.001$, t-test) with the integration of the 10-day-period of measurements (29 August 2005–6 September 2005) yielding a difference of 18 mm of evaporated water in favour of the grassland.

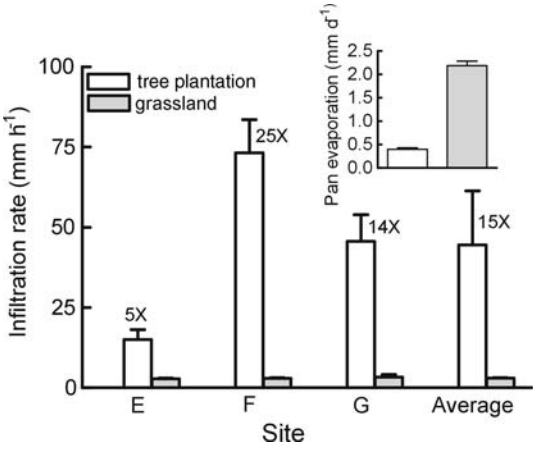

Fig. 6 Water infiltration and pan evaporation rates under native grassland and oak plantations pairs (mean ± SE). Infiltration rates (mm h^{-1}) were measured in three pairs (sites E–G) using a double-ring infiltrometer ($n = 3$ per stand). Pan evaporation rates (mm day^{-1}) were measured in site E ($n = 3$ per stand) using micro-evaporimeters (10 cm diameter) for 10 days (29 August 2005–6 September 2005). Differences were significant in all cases ($P < 0.05$)

A depression of the water table in the tree plantation with respect to the adjacent grassland together with a clear pattern of diurnal fluctuations suggested groundwater use by trees at site E (Fig. 7). Differences in the absolute groundwater levels (GWL) measured on September 2005 approached 26 cm ($GWL_{plantation}$ = 504 cm, $GWL_{grassland}$ 1 = 478 cm, $GWL_{grassland}$ 2 = 480 cm) indicating the presence of a depression cone under the plantation. Permanent depth recording at two of these wells indicated that this depression increased to >60 cm at the end of the summer (a larger depression may have taken place since the water table declined below the bottom of the plantation well for a few days at this time) and showed a steady reduction after leaf fall (Fig. 7).

Diurnal watertable fluctuations of at least 10 mm were observed for 87% of the 46-day period before the onset of leaf fall, indicating a direct discharge of groundwater by trees (Fig. 7). In this period, the average fluctuation approached 55 mm. Daily watertable decline started at 0830–0930 hours, likely corresponding with transpirational uptake by oak trees, while the rise in the watertable started at 1730–1830 hours, suggesting the dominance of hydraulic equilibration fluxes once transpiration withdrawals ceased. Rainfall events interrupted fluctuations and caused a continuous rise in the groundwater level both in the night and daytime in the plantation but not in the grassland, indicating a cessation of groundwater use by trees (Fig. 7). At the end of the measurement period, when leaves had almost completely fallen, fluctuations disappeared.

The chloride mass balance and the water table fluctuation analysis provided convergent estimates of groundwater consumption by oak trees at site E. Considering the whole soil column down to 6 m depth (at this depth chloride concentrations in both stands converged), the tree plantation displayed a net chloride gain of 3.06 kg chloride m^{-2} or approximately 71.1 g chloride m^{-2} $year^{-1}$. With an average groundwater chloride concentration of 183 mg l^{-1} at 6 m depth, an average annual groundwater uptake of ~388 mm $year^{-1}$ would have been needed to convey the mass of chloride accumulated in the plantation. Diurnal water table fluctuations suggested that groundwater use by vegetation averaged 1.87 mm day^{-1} (range 0.55–

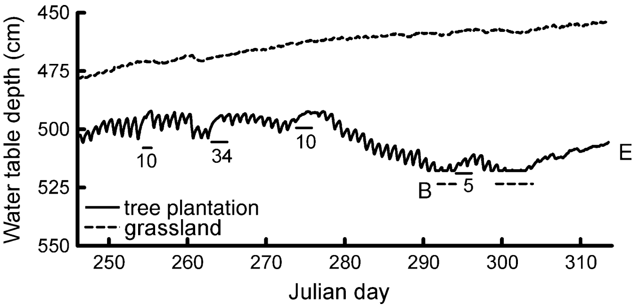

Fig. 7 Water table dynamics under native grassland and oak plantation at site E. Water table levels were measured from 30 August 2005 to 9 November 2005 at permanent monitoring wells located 40 m from the plantation–grassland edge (see Fig. 1). Levels correspond to the absolute elevation of the water table with respect to the ground level. Water levels were monitored at high vertical (<3 mm) and temporal (30 min) resolution. Rainfall events registered during the monitoring period are indicated by *solid lines* and the rainfall amounts are indicated *below*. The beginning (*B*) and end (*E*) of the leaf fall are indicated. *Dotted lines* indicate periods in which the plantation well became dry

3.19 mm day^{-1}) during the study period prior to the leaf fall stage. Taking into account that trees have leaves only for 6 months, a rough annual integration of these measurements would give a value of 341 mm year^{-1}. These independent estimates showed good agreement, suggesting that groundwater contributions may complement precipitation (540 mm year^{-1}) raising water inputs to tree plantations by two-thirds.

Discussion

The establishment of oak trees on natural salt-affected grasslands of the Great Hungarian Plain has substantially changed water dynamics at different scales, altering salt transport/storage patterns along the ecosystem–vadose zone–aquifer continuum. Water flow contrasts between grasslands and plantations left their imprint on salt distribution through the flushing of surface soil salts accumulated before hydrological regulation and the development of a new and stronger salt accumulation zone at deeper soil layers.

By modifying vertical water movement, grassland afforestation triggered the desalinization and dealkalization of shallow soil layers, something that the earlier lowering of the water table was unable to accomplish on its own. Although more than a century has passed since the intervention of humans in the regional hydrology of Hortobagy decoupled the salt source (i.e. groundwater) from the soil surface of most of its native ecosystems, the grasslands still present large salt accumulations in their upper soil layers (Ábrahám and Bocskai 1971; Figs. 3, 4). The vertical distribution of salts in grassland soils, with a bulge at ~50–150 cm, suggests that they date from pre-drainage times when the water table was shallower. Likely, these salts remained unflushed because the high sodium content and fine soil texture interacted severely hindering the downward movement of water and salts (Schofield et al. 2001; Szabolcs 1989; Figs. 2, 4, 6). However, the removal of this salt bulge following tree establishment was confirmed in the seven oak plantation–grassland pairs (Fig. 3). This result, together with our infiltration measurements and the observation of pond formation in grassland stands but not in plantation stands following a large rain event (60 mm on 22–29 August 2005), supported the hypothesis that trees favoured salt leaching in these soils through changes to their physical properties. These contrasts in the downward movement of water match other observations which have compared trees/shrubs and grassland sites (Deuchards et al. 1999; Devitt and Smith 2002; Eldridge and Freudenberger 2005) and may result from a higher root density (Jackson et al. 1996), microfaunal activity (Munzbergova and Ward 2002), and/or higher content of soil organic matter and litter (Grieve 1980) of afforested sites improving the porosity and water transport capacity of soils. The leaching of soluble salts that commonly follows the afforestation of sodic soils in the Caspian region and India (Mishra and Sharma 2003; Sizemskaya and Romanenkov 1992) would also be explained by the mechanisms observed in Hortobagy.

Although the conversion of grassland to plantation has clearly promoted salt removal in upper soil layers, the opposite took place at greater depth, with salts accumulating in the vadose zone and aquifer (Fig. 4). Net salt gains in tree plantations have been likely triggered by the re-initiation of the groundwater discharge regime that was interrupted ~150 years ago following regional artificial drainage. Differences between the original salinization process and the one triggered by tree plantations should be pointed out, however. While salinization in the native grasslands took place in the upper soil layers and was accompanied by soil alkalinization (Schofield et al. 2001, Fig. 4), the new, tree-mediated, salinization process proceeded at greater depth, in the capillary fringe zone, and was not accompanied by alkalinization (similar soil pH and lack of significant sodium gains; Fig. 4). In addition, trees promoted a more intense salt accumulation process in comparison with the original grasslands, as suggested by the sharp chloride bulge that they generated (Fig. 4). This contrast probably resulted from the greater evaporative capacity that tree canopies display in comparison with grasslands as a consequence of their higher aerodynamic conductance (Calder 1998; Kelliher et al. 1993).

Vegetation shifts, interacting with human interventions, led to profound changes in the regional hydrology of Hortobagy that translated into new and contrasting salt accumulation and distribution patterns. Before hydrological regulation, the poor drainage network of the region determined the presence of a shallow water table across most of the landscape, favouring a discharge regime in which grass transpiration together with soil and pond evaporation balanced precipitation inputs leading to surface salt accumulation (Schofield et al. 2001; Fig. 8a). With the hydrological regulation process, the regional drainage network was significantly improved and the water table lowered, decoupling it from the surface ecosystem (Szabolcs 1979; Fig. 8b). Although evaporative groundwater discharge was interrupted and, as a consequence, salt accumulation stopped, the existing salt pool remained unflushed due to the poor physical condition of soils. After tree establishment, downward water percolation was significantly increased and as a consequence, salts in the upper soil layers were flushed and ponding and overland flow eliminated (Fig. 8c). At greater depth, however, tree roots depressed the water table through groundwater absorption, generating Darcian flows from surrounding grasslands that supplied water and salts to the plantation. An evaporative groundwater discharge regime in which transpiration (as opposed to soil and pond evaporation) was the most important component was established and a new salt accumulation process began.

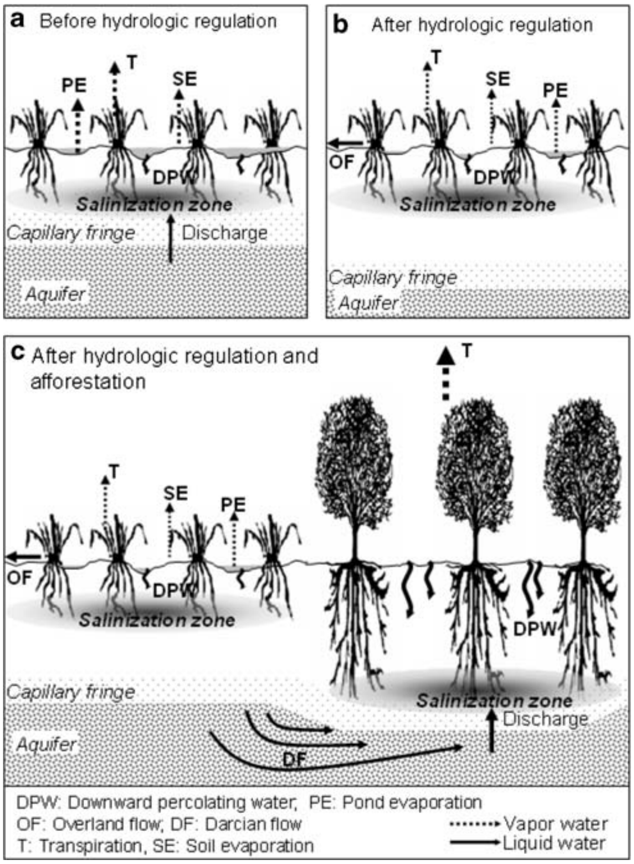

Fig. 8a–c Hypothetical mechanisms shaping the salinization patterns observed at grassland and tree plantations of Hortobagy. Before hydrological regulation the water table was shallow enough to allow a discharge regime sustained by transpiration, and soil and pond evaporation, leading to salt accumulation in the upper soil layers (**a**). After hydrological regulation, the water table was lowered and the discharge regime interrupted, but salts remained due to poor soil physical conditions preventing their leaching (**b**). With afforestation salts were flushed from the upper soil layers but the discharge regime was re-established leading to a deeper and stronger salt accumulation (**c**)

The establishment of trees on grassland ecosystems by means of both human-induced afforestation and natural invasion is taking place around the world at increasing rates (Geary 2001; Jackson et al. 2002) and may intensify in the near future motivated by an eventual carbon sequestration market (Wright et al. 2000). Beneficial and problematic consequences could emerge from this land use shift. We have shown that trees, by changing soil physical properties, may enhance water infiltration and help to reclaim naturally salt-affected soils. At the same time, large-scale afforestation can trigger a substantial draw-down of water tables that could be beneficial in places where shallow water tables constrain crop production (e.g. Australia, Barret-Lennard 2002; Argentina, Taboada and Lavado 1988); but deleterious if the higher water use of trees (Nosetto et al. 2005) translates into lower water yield (Farley et al. 2005) and impaired groundwater renewal (Jobbágy and Jackson 2004), eventually jeopardizing other water uses. Moreover, groundwater use by trees may enhance forest productivity, but at the same time, trigger a rapid secondary salinization of vadose zones and aquifers compromising the sustainability of forestry in the long term.

Besides proving that vegetation changes can have strong effects on water dynamics with cascading consequences on salt accumulation and distribution, our study has shown that focusing on the whole ecosystem–vadose zone–aquifer continuum is critical to understanding salinization processes operating at different temporal and spatial scales. Interestingly, if we had focused on the surface ecosystem (up to 1 m depth of the soil profile), as in most ecological studies, we would have concluded that afforestation promotes the desalinization of the ecosystem, far from the real situation. Furthermore, tools commonly used in hydrological studies, such as salt balances and water table fluctuations analysis, could supplement ecological techniques, allowing a better understanding of ecosystem process. A broad ecohydrological perspective that explicitly considers the influence of vegetation on groundwater hydrology is needed to anticipate the consequences of land use changes on water and soil resources.

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